Aquatic Macrophyte Response to Carp Removal

and the

Success of Transplanting Aquatic Macrophytes

to Restore the Littoral Community

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Dedication

This thesis is dedicated to my wife Melissa for her patience, unconditional support, and personal sacrifices made on my behalf throughout these past few years.

Comprehensive Abstract

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health. They provide refuge for algae-consuming zooplankton, stabilize sediment, consume nutrients, and provide habitat for aquatic vertebrates. However, high densities of non-native invasive macrophytes can cause harm to the lake ecosystem. A healthy aquatic macrophyte community consists of a diversity of native taxa. Common carp, an invasive species, can disrupt native macrophyte communities. High densities of common carp uproot aquatic macrophytes and stir up sediments, encouraging a turbid water state and low density of macrophytes. Water clarity and aquatic macrophytes will likely increase after the removal of common carp. However, it can take several years for native macrophytes to return after a large-scale fish manipulation. There is also concern that non-native invasive macrophytes will quickly dominate the littoral zone after carp removal.

I evaluated the response of the aquatic macrophyte community, between 2009 and 2012, after the large scale removal of common carp from Lake Susan, Carver County, MN (MN DNR DOW ID 10-001300). To assess plant frequency of occurrence, plant communities were assessed using a point intercept survey method in the spring and mid-summer. To quantify changes in plant biomass, approximately 40 random biomass samples were taken during each survey. To further understand response of Eurasian watermilfoil (*Myriophyllum spicatum*), a non-native invasive plant, milfoil herbivore surveys were also conducted throughout each summer. To evaluate recruitment potential of the non-native curlyleaf pondweed (*Potamogeton*

crispus) turions were sampled each October. Comparisons of the plant community were done between the years to understand the succession of natural plant recruitment.

To promote the recruitment of native plants, I transplanted six native taxa from nearby Lake Ann into Lake Susan to increase the species richness and distribution macrophytes within the lake. Four separate transplant experiments were conducted, each assessing a different variable. The primary focus of experiment one was to assess the survival of selected taxa and compare growth between open locations and those protected from herbivore access. The primary focus of experiment two was to increase the number of transplant locations geographically around the littoral zone, and correlate environmental growth factors (light, sediment bound nitrogen, sediment bulk density, sediment organic matter, and wave protection behind lily beds), to the expansion of the transplanted taxa. Experiment three tested transplant survival and expansion at greater depths. Experiment four evaluated if earlier planting of transplants improved survival of deeper transplants.

There was an increase in the diversity and overall abundance of submersed aquatic macrophytes in Lake Susan after common carp population was reduced. Species richness increased naturally from 13 taxa in 2009 to 14 in 2012. The number of sites with moderate diversity (>4 taxa per site) increased from 8 sites to 2009 to 12 sites in 2012. There was also an increase in the frequency of occurrence of most taxa. Total dry shoot biomass of both native and invasive taxa also showed a statistically significant increase ($p < 0.001$). The non-native curlyleaf pondweed increased significantly from 17% of sites in 2009 to 41% of sites by 2011. Eurasian watermilfoil decreased in frequency and biomass between 2009 and 2012. This decline, with the persistent high density of milfoil weevils suggests the weevils provided effective biocontrol of Eurasian watermilfoil.

Transplanting whole adult plants in shallow water (<1 meter) was generally successful for the duration of this study (four years). Most transplanted taxa showed survival during the initial growing season. Overwintering success was a better predictor of long term success than initial survival and protection from by lily beds. Wild celery (*Vallisnaria americana*), and water-stargrass (*Zosterella dubia*) had the highest survival, whereas bushy pondweed (*Najas flexilis*), and muskgrass (*Chara spp.*), had inconsistent survival, and northern watermilfoil (*Myriophyllum sibiricum*), largely failed to survive. The most important environmental factor in the success and expansion of transplants was depth, which affected light availability. Whole plants transplanted into deeper zones (approximately 1.4m depth) failed due to low light (summer Secchi depths were often <1.0m). Both native and non-native aquatic macrophytes responded favorably to the removal of high carp densities, and transplanting aquatic macrophytes can help to restore the littoral community.

Table of Contents

List of Tables

List of Figures

Prologue

In this thesis there are four chapters. Chapter I is a brief overview of the role of native aquatic vegetation in lake ecosystems and implications of high densities of benthivorous fish. Chapter II evaluates the natural response of the aquatic macrophyte community after the removal of common carp (*Cyprinus carpio*) from a eutrophic lake, Lake Susan, Carver County, Minnesota, USA. The majority of common carp were removed from Lake Susan in 2008-2009. After the removal of carp, the distribution, abundance and composition of the submerged aquatic macrophyte community was assessed over four years (2009-2012). The primary objective of this chapter was to monitor the natural succession of submerged aquatic macrophytes after the large scale removal of benthivorous fish.

Chapter III evaluates the survival and expansion of transplanted submersed aquatic macrophytes as a means for improving the aquatic plant cover, biomass, and species composition in the littoral zone. This chapter consists of four separate experiments in which a total of six native submerged aquatic macrophyte taxa were collected from a nearby lake and transplanted into Lake Susan.

Chapter IV summarizes the overall thesis and ties together chapters II and II. The overall goal of this project was to gain a deeper understanding of the dynamics involved in the restoration of an impaired lake. Gaining a deeper understanding of the link between the role of carp, aquatic macrophytes, and lake ecosystem health will further our knowledge of lake ecosystem restoration.

Chapter I

Overview of Restoration of Submersed Aquatic Macrophytes

Background

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health. Aquatic vegetation provides habitat for zooplankton, which feed on algae, thus providing increased water clarity (Valley et al. 2004, Norlin et al. 2005). Aquatic macrophytes also play a crucial role in decreasing sediment and phosphorus re-suspension (Horppila and Nurminen 2003, Sondergaard et al. 2003). Many taxa of submersed aquatic macrophytes have also been shown to directly compete with pelagic algae for nutrients in the water column (van Donk 2002). By adding to structural complexity within littoral zones, aquatic macrophytes provide habitat for epiphytic organisms, increasing the abundance and diversity of aquatic life (Carpenter and Lodge 1986). For example, Shoup et al. (2003) demonstrated increased survival of sunfish (*Lepomis spp*.) in submersed aquatic vegetation, due to increased foraging opportunities and evasion from predators. The combination of directly reducing re-suspension of nutrient laden sediment, competing with algae for the uptake of suspended nutrients, and providing structure for epiphytes and zooplankton creates a positive feedback loop to help maintain water clarity (Carpenter and Lodge 1986, Scheffer et al. 2001).

The common carp, being benthic feeding fish, stir up sediment and uproot vegetation (Crivelli 1983, Hanson and Butler 1994, Scheffer 1998). Roberts et al., (1995) found that muskgrass and wild celery were especially vulnerable to carp uprooting. Reduced vegetation increases the available wave energy in shallow regions, which can further increase sediment re-suspension. Sediment re-suspension increases turbidity and releases sediment bound nutrients, which in turn feed algae blooms (Scheffer 1998). The increase in suspended sediment and algae decreases the water clarity and available light for rooted vegetation, further driving the turbid water state (Hanson and Butler, 1994). Shallow lakes with high carp abundance and few aquatic macrophytes also attract few waterfowl (Bouffard and Hanson 1997, Hass et al. 2007). Reducing the carp population in turbid lakes with very high abundance of common carp can be a means of restoring the aquatic macrophytes and improving water quality (Schrage and Downing 2004, Bajer et al. 2009).

Removing the majority of common carp from a turbid lake with high carp densities will reduce uprooting and likely lead to an increase in water clarity and aquatic macrophytes (Schrage and Downing 2004, Bajer et al. 2009). The increase in water clarity may have a positive effect on growth of both native and exotic submersed macrophyte species. Of particular concern is the response of invasive macrophytes. Lauridsen et al., (1994) found that invasive macrophytes became the dominate species four years after a manipulation of fish was executed in a shallow turbid lake. High densities of invasive macrophytes have been shown to reduce fish growth (Olson et al. 1998). Invasive macrophytes that form mats on the surface also become a nuisance for recreational and commercial water users (Aiken et al. 1979, Smith and Barko 1990). A diverse population of native plants however, can help to reduce the invasive nature of some exotic species (Madsen 1997), with the added benefit of providing structural complexity to the littoral zone.

Once high populations of benthivorous fish, such and carp, are removed from the lake sediment re-suspension should decrease and water clarity should improve. However, it can take several years for aquatic macrophytes to return in high densities after fish removal (Hanson and Butler 1994, Hanson et al. 2006, Lauridsen et al. 2003). Thus manipulation of the littoral community may be necessary to promote the establishment native vegetation rather than invasive taxa.

There is a large body of information on aquatic plant management and development of various methods for encouraging growth of native submersed macrophytes while inhibiting the growth of exotic invasive species. Most studies evaluate the use of chemical or mechanical controls, or altering hydrologic regimes to manipulate submersed aquatic plant communities (Madsen 1997) to favor the growth of natives. There are relatively few published papers that specifically pertain to transplanting submersed aquatic vegetation as a means for increasing habitat, biomass or diversity of native species. However, since the early 1990’s there has been an increasing trend to use planting as a means to manipulate the species composition within the littoral zone of lakes. Transplanting has been found to be an effective method for introducing submersed aquatic macrophytes into water bodies where few propagules exist (Smart et al. 1998, Lauridsen et al. 2003, Smiley and Dibble 2006). Transplanting has also been done in lakes to restore the plant community after a lake wide biomanipulation of the fish community (Lauridsen et al. 2003).

Although whole plant transplanting can be effective, it is also time consuming and potentially expensive (Smart et al. 2005). Smart et al. (1996) pioneered much of the early work of determining which taxa make successful candidates for transplanting and established recommended methodology. They determined effectiveness of transplanting ten taxa by tuber, bare root, or peat pots. They identified muskgrass (*Chara vulgaris),* bushy pondweed (*Najas guadalupensis*)*,* narrowleaf pondweed (*Potamogeton pusillus*)*,* horned pondweed (*Zannichellia palustris*)*,* longleaf pondweed (*P. nodosus*)*,* sago pondweed (*Stuckenia pectinata*)*,* wild celery (*Vallisneria americana*)*,* Canada waterweed (*Elodea canadensis*)*,* and water stargrass (*Zosterella dubia*) as strongly recommended taxa for establishment in reservoirs, particularly in the southern United States.

Although many of the initial concepts and procedures for transplanting submersed aquatic macrophytes have been developed (Smart et al. 1998, Smart et al. 2005, Fleming et al. 2011), the majority of case studies have been carried out in the southern United States and temperate climates of Europe (Dick et al. 2004a, Dick et al. 2004b, Flemming et al. 2011, Lauridsen et al. 2003). With the exception of Storch et al., (1986) and arguably Lauridsen et al., (2003) few transplanting case studies have been done in northern latitudes.

Lauridsen et al., (2003) transplanted three *Potamogeton* taxa in Lake Engelsholm, Denmark, when aquatic macrophytes failed to re-establish after two years of a large scale fish biomanipulation. Although they were able to get the plants to establish and colonize in the lake, grazing by waterfowl, even when at low densities, was shown to negatively affect success. The effects of grazing waterfowl were dramatically reduced with the use of a protective mesh barrier. Although the transplants were replicated around the lake, there was no record of expansion over time, in part due to herbivory outside of the protected enclosures (Lauridsen et al. 2003). More recent work done by Fleming et al., (2011) involved testing transplant success rates in situ with three taxa and three depth ranges (0.3m, 0.6m and 1.0m). They concluded planting depth (up to 1.0m) had no effect on survival rate, but species selection did have an effect on survival rate.

Little reference was given in the previously mentioned studies to the expansion over time of the transplants. Some early work by Storch et al., (1986) had reasonable success transplanting large leaf pondweed, *Potamogeton amplifolius,* in a western New York Lake. Overall success rate (percent of original plants still alive after 3-4 years) was good, but the expansion of the transplants (about $1m^2$) was poor. Expansion was addressed to some degree in Dick et al. (2004a,b) in which area of growth of the most successful species were measured, but only over one growing season.

There are several environmental factors related to the growth and expansion of the transplants, such as depth, light attenuation (Madsen and Sand-Jensen 1994), sediment texture, (Sculthorpe 1967, Smart et al. 1996), disturbance frequencies (Barrat-Segretain, et al. 1999), sediment organic matter and available nitrogen, (Barko and Smart 1983, Wijck et al. 1992). Comparing these environmental factors to survival and growth may suggest which factors lead to future successful transplanting. Documenting the survival and expansion of transplanted submersed aquatic macrophytes will assist the planning and development of future littoral vegetation habitat restoration projects.

Factors affecting macrophyte growth and colonization

Light as a factor for aquatic macrophytes growth

The distribution of macrophyte taxa, regionally as well as within the lake basin, is related to a number of environmental factors. One of the most important environmental factors for the growth of submersed macrophytes is light availability (Scheffer 1998). Best et al., (2001) describes three important considerations when looking at light in the water column. Irradiance influences the production of biomass, reproduction and morphology; wavelength influences morphologenetic processes; and day length can trigger phenological events. Maximum depth of colonization of submersed aquatic vegetation can generally be predicted by transparency as there is a positive correlation between maximum depth of colonization and Secchi disk depth (Canfield et al. 1995). Light compensation point, which determines the maximum depth of growth for individual species, also varies between species (Barko and Smith 1981). Shade tolerance allows some taxa to have a competitive advantage in light limited conditions, ultimately influencing the composition of the aquatic plant community. Light conditions can also influence morphological growth of submersed aquatic vegetation. In low light conditions, many submersed aquatic plant species, especially invasives like *M. spicatum*, put more energy into shoot length and biomass in the development of canopy formation and shoot growth (Barko and Smart 1981).

Temperature as a factor on plant growth

Temperature is another important factor in the growth of aquatic plants. Temperature's influence on aquatic plants begins with triggering germination of seeds, tubers, and over wintering buds. Spencer and Ksander (1992) demonstrated that sediment temperature, not innate dormancy; trigger germination of vegetated structures, which influences the phenology of many *Potamogeton* species. It can be difficult to completely distinguish the influence of temperature from that of light, because solar radiation acts to warm water. Temperature was found to be highly influential to total plant biomass production, with an increase in temperature relating to an increase in biomass up to a critical temperature (Barko and Smart 1981). Latitude, seasonal temperature and species range of thermal tolerance appear to be important drivers of geographical distribution of submersed macrophyte taxa, whereas temperature appears to be highly influential in seasonal growth cycles and range in latitude of colonization.

Wave energy as a factor influencing growth

Wave energy likely plays a role in the ability of submersed vegetation to grow and expand (Doyle 2001). One of the largest indirect roles of wave energy is substrate composition related to high energy environments Madsen et al., (2001). As wave energy increases so too does the particle sizes remaining. High wave energy littoral areas include areas of the shoreline with a large fetch, or narrow areas with high volumes of boat traffic. Plant growth has been shown to be reduced in areas with even mild wave action. In areas with high wave energy, there is a reduction in fine sediments and often an increase in turbidity, resulting in few to no plants that may be able to establish or survive. In a controlled experiment, Doyle (2001) found that *Vallisneria americana* biomass was 50% lower than the control after being exposed to even light wave action (0.15m height) 5-6 times per day over a 67 day period.

Several previous transplant case studies have noted that protection of transplants is an important factor to consider (Smart et al. 1996). Lauridsen et al., (1993) found that fenced-in transplants grew 6.5 times more than transplants at non-fenced locations and that a combination of wave action and herbivory from waterfowl were factors. Wave barriers were also an important factor in successful establishment of the emergent species *Schoenoplectus tabernaemontani* (Vanderbosch and Galatowitsch 2011). In each of the noted examples anthropogenic barriers, usually in the form of fences, were used. Little work has been done to determine if stands of floating leaf vegetation are useful in providing enough wave protection to transplants.

Sediment texture and organic matter as a factor on plant growth

The growth of submersed aquatic plants, like that of terrestrial plants, is in part influenced by the composition of the substrate in which they grow (Sculthorpe 1967). Sculthorpe (1967) emphasizes that physical texture is likely more important than soil nutrient composition. However, Barko and Smart (1983) suggest there is little evidence for this, and points to sediment organic matter as a more likely limiting factor to aquatic plant growth than sediment texture. In laboratory experiments, Barko and Smart (1983) found that biomass of plants increased as sediment organic matter increased, up to about 20% organic matter, after which levels inhibited the growth of submersed aquatic plants. However, there was increased variability in biomass at very low levels of organic matter. Barko and Smart (1986) concluded that the combination of sediment organic matter and sediment texture are important in limiting the growth of aquatic vegetation.

Available nitrogen as a factor on plant growth:

Nutrition plays an important role in the success of aquatic macrophytes. Barko et al., (1991) points out that submerged macrophytes can uptake nutrients, including nitrogen, from both root structures in the sediment and directly from the water column. Sediment-bound nutrients are closely related to changes in the aquatic macrophyte community. Submersed aquatic macrophytes primarily rely on sediment as the source for the uptake of nitrogen and phosphorus (Barko et al. 1991). Due to diagenic processes, pore water sediment can have higher concentrations of nutrients than overlaying water (Bufflap and Allen 1994). Nitrogen is depleted from sediments more rapidly than phosphorus; hence nitrogen is more likely to become a limiting factor to macrophyte growth (Barko et al. 1991). The sediment nitrogen availability may play an important role in the success and growth of transplanted taxa. To understand which factors play a critical role success of submersed aquatic macrophytes, numerous environmental variables need to be considered.

Planting time as a factor on establishment:

Timing of transplants was found to be one of the most important variables that lead to successful establishment of the emergent macrophyte *Schoenoplectus tabernaemontani* in central Minnesota (Vanderbosch and Galatowitsch 2011). Timing can influence the success rate of transplant propagules in several different ways. Temperature of the sediment and water column, especially in northern latitudes, is dependent on timing. Life stage and plant maturity are primary factors related to the timing of whole plant propagules (Vanderbosch and Galatowitsch 2011).

Summary

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health. They provide habitat for zooplankton, fish, and other wildlife such as waterfowl (Valley et al. 2004, Norlin et al. 2005). They also help maintain water quality by reducing suspension of sediment and competing with algae for nutrients (Horppila and Nurminen 2003). High densities of common carp decrease aquatic macrophytes and impair water quality and ecosystem health. Reducing the population of common carp has been shown to increase the macrophyte density and improve water quality (Bajer et al. 2009).

It can take several years for aquatic macrophytes to return in high densities after fish removal (Hanson and Butler 1994, Hanson et al. 2006, Lauridsen et al. 2003). Lauridsen et al., (2003) found that only four years after fish population manipulation, non-native invasive macrophytes became dominant. High densities of non-native invasive macrophytes can also impair ecosystem health and recreation opportunities (Olson et al. 1998, Smith and Barko 1990). Thus manipulation of the littoral community may be necessary to promote the establishment native vegetation rather than invasive taxa. Whole plant transplants can be an effective means to increase the native plant diversity in the littoral zone (Smart et al. 2005).

The distribution of aquatic macrophytes is related to a number of environmental factors. Light availability is one of the most important factors (Scheffer 1998), as it influences the depth of growth and taxa diversity. Temperature is also an important factor as it can trigger germination, impact seasonal growth cycles and geographic distribution (Spencer and Ksander 1992, Barko and Smart 1981). Sediment texture and organic matter content influence the density and distribution of macrophyte growth (Barko and Smart 1986). Wave energy influences the density of macrophyte growth and the sediment structure (Doyle 2001). Rooted aquatic macrophytes uptake nutrients, such as nitrogen, from the sediment (Barko et al. 1991); thus sediment-bound nutrients are closely related to changes in the aquatic macrophyte community. Timing also influences the success rate of transplanting propagules. (Vanderbosch and Galatowitsch 2011).

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Chapter II

Aquatic Macrophyte Population Response to Large Scale Removal of Common Carp

Summary

I evaluated the response of the aquatic macrophyte community after the large scale removal of common carp in 2009 from Lake Susan, Carver County, MN (MN DNR DOW ID 10-001300), with surveys in 2009-2012. The frequency of occurrence and changes in plant biomass were assessed using a point intercept survey method in the spring and mid-summer. The milfoil weevil population was assessed to better understand changes in Eurasian watermilfoil (*Myriophyllum spicatum*), a non-native invasive. Turions were evaluated to understand recruitment pressure of curlyleaf pondweed (*Potamogeton crispus*), another non-native invasive. Comparisons of the plant community were done between the years to understand the succession of natural plant recruitment.

There was an increase in the diversity and overall abundance of both native and non-native submersed aquatic macrophytes in Lake Susan after the reduction of common carp abundance. Species richness increased naturally from 13 taxa in 2009 to 14 in 2012. The number of sample sites with moderate plant diversity (>4 taxa per site) increased from 8 sites in 2009 to 12 sites in 2012. There was also an increase in the frequency of occurrence of most taxa. Total dry shoot biomass of both native and invasive taxa also showed a significant increase ($p<0.001$). However the maximum depth of plant growth largely remained unchanged, about 4.0m, across four years. Eurasian watermilfoil, a non-native species, decreased in frequency and biomass between 2009 and 2012. Milfoil weevil population stayed fairly high (about 0.1 weevils/stem) throughout the same time period. The non-native species curlyleaf pondweed, increased significantly from 17% of

sites in 2009 to 41% of sites by 2011. After the reduction in carp abundance the native plant population increased, but strategies to contain invasive species are needed.

Introduction

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health. They provide habitat for zooplankton, fish, and other aquatic wildlife. Aquatic macrophytes help maintain water clarity by harboring zooplankton that feed on algae, (Valley et al. 2004, Norlin et al. 2005), and by decreasing sediment and phosphorus re-suspension (Horppila and Nurminen 2003, Sondergaard et al. 2003). Submersed aquatic macrophytes have also been shown to directly compete with pelagic algae for nutrients in the water column (van Donk 2002). The combination of directly reducing re-suspension of nutrient laden sediment, competing with algae for the uptake of suspended nutrients, and providing structure for epiphytes and zooplankton creates a positive feedback loop to help maintain water clarity (Carpenter and Lodge 1986, Scheffer et al. 2001).

High densities of benthic feeding fish, such as the common carp, stir up sediment and uproot aquatic macrophytes reducing water clarity (Crivelli 1983, Hanson and Butler 1994, Scheffer 1998). Shallow lakes with high carp abundance and few aquatic macrophytes also attract few waterfowl (Bouffard and Hanson 1997, Hass et al. 2007). Greatly reducing the density of common carp from turbid lakes with high carp densities can result in increased water clarity and submersed aquatic macrophytes (Schrage and Downing 2004, Bajer et al. 2009). One factor that may determine the macrophyte species community that develops after removal of high carp densities is the composition of the seed propagule bank in the sediment (Hilt et al. 2006). Another factor is the recruitment strategy of the few species that are present in the lake. For example, taxa that have aggressive recruitment tendencies, such as non-native invasives, will likely become early colonizers (Lauridsen et al. 1994). Of particular concern is the presence of non-native invasive species. High densities of invasive macrophytes cause ecological problems, such as out-competing native taxa, reducing fish growth, and forming mats on the surface that become a nuisance for recreational and commercial water users (Olson et al. 1998, Aiken et al. 1979, Smith and Barko 1990).

In this study lake, (Lake Susan, Carver County, MN (Minnesota Department of Natural Resources (MN DNR), Division of Waters (DOW) ID 10-001300), two non-native invasive taxa, curlyleaf pondweed (*Potamogeton crispus*) and Eurasian watermilfoil (*Myriophyllum spicatum*), were already established in the lake, albeit at low densities due to the high carp population. Aggressively colonizing non-native species are a concern because they can quickly become the dominate species and create an unbalanced monotypic littoral plant community (Lauridsen et al. 1994) after the reduction of carp density. A diverse population of native plants however, can help to reduce the invasive nature of some non-native species (Madsen 1997), with the added benefit of providing structural complexity to the littoral zone.

Curlyleaf pondweed is an invasive submerged aquatic macrophyte to North America (Bolduan et al. 1994). This species can form dense mats on the water surface causing direct competition with and shading of native species. Curlyleaf pondweed can out-compete native plants due to its ability to grow well in cold temperatures (Kunii

1982). Curlyleaf pondweed turions (vegetative overwintering buds) often sprout in the late fall and grow slowly over the winter when native plants are typically dormant. In early spring, curlyleaf then grows rapidly and forms a dense canopy at the surface where it out competes the native plants for light (Bolduan et al. 1994). Although curlyleaf pondweed can reproduce sexually via seed, vegetative reproduction by turions is usually the most common. Thus, assessing the turion bank in the substrate during the fall can be a useful method for predicting the density and spatial extent of curlyleaf pondweed the next spring (Bolduan et al. 1994).

Eurasian watermilfoil, introduced to North America from Eurasia, can also become a nuisance by forming large monotypic stands that create dense mats on the surface (Smith and Barko 1990). The crowding and decreased light below the canopy inhibits the growth of native taxa. Dense monotypic stands have been shown to reduce fish growth and decrease recreation (Olson et al. 1998). Eurasian watermilfoil can reproduce by seed and by vegetative fragments that break off from the surface canopy, float to other areas of the lake, sink and re-grow (Smith and Barko 1990). This form of vegetative fragmentation allows Eurasian watermilfoil to expand rapidly becoming a lake-wide issue.

The purpose of this case study was to quantify the native and non-native submersed macrophyte community that naturally recruited after the majority of carp were removed from Lake Susan. The frequency of occurrence, dry plant biomass, spatial zonation and succession of the plant community was quantified and compared over the four year study period.

Methods

Study Area

The study area, Lake Susan, Carver County (MN DNR), ID 10-001300), is a eutrophic lake in central, Minnesota, USA. The lake covers about 38 hectares, with approximately 30 hectares shallower than 4.6m in depth (MN DNR defined littoral zone) and a maximum depth of 5.2m (MN DNR Lake finder 2011). Normal annual water level fluctuations are ± 0.25m. Prior to carp removal, Lake Susan had low summer Secchi depths, few submerged aquatic plants, and a high abundance of common carp. In winter (March) 2009, approximately 78% of the carp were removed from the lake (from 307kg/ha in 2008, to 65kg/ha in 2009) to determine if reduction of carp would enhance water clarity and improve the native plant community (Bajer and Sorensen 2012). Lake Susan is polymictic and appears to have high internal phosphorous loading when the hypolimnion goes anoxic, as shown by the significant increases in chlorophyll a and total phosphorous, resulting in low Secchi disk (MPCA 2012) values in summer (Table 1).

Assessment of Aquatic Plant Community

The frequency of occurrence of aquatic macrophytes was assessed using point intercept surveys following protocols established by Madsen (1999). The surveys were performed in spring and summer of 2009, 2010, 2011, and 2012. The spring vegetation survey (May-June) was timed to coincide with the peak biomass of the non-native species curlyleaf pondweed (*Potamogeton crispus*,) and the summer survey (July-August) to coincide with peak biomass of native species.

ArcMap GIS software was used to generate survey points following a systematic square grid. Grid spacing of 50m produced 146 point intercept sampling points and ensured at least 120 sampling points were within the littoral zone (≤4.6m depth). The same sampling points were used in each survey to allow for consistent comparisons between surveys. The sampling points were loaded into a Garmin GPS and a boat was navigated (within 5 meters) to each sampling point. A weighted double headed rake (0.33m wide) attached to a rope was then tossed into the lake, allowed to sink and dragged along the lake bottom for approximately three meters, thus sampling approximately one square meter. The vegetation collected was identified and a semi-quantitative rake density rating, 0 to 5, was visually estimated for each taxon present. For this density rating, 1 was noted for a single stem, 2 was noted for a few stems, 3 for many stems (about ½ rake full), 4 indicates a full rake, and 5 was noted for a full rake with very dense surface mats. This rake rating quantification was used to determine statistical differences between taxa and years. Crow and Hellquist (2000) was used as taxonomic authority. Frequency of occurrence was determined for all native plants and all invasive plants within the littoral zone. Species richness was determined as the total number of taxa present at each sampling point. Taxa found near a sampling point but not collected in the rake sample were noted and added to the total number of species found in the lake, but not used in any calculations. ArcMap GIS software was used to generate maps to assist in visualizing plant taxa locations, depth of growth, and richness at sampling points.

Dry plant shoot biomass was assessed with the vertical rake method (Johnson and Newman 2011). Biomass was sampled at 40 sites randomly selected from the point intercept sampling points. The same sampling locations (±5m) were used in each survey to provide more consistent comparisons among years. An extended-handle garden rake (0.33m in width) was lowered to the lake bottom, rotated three times to ensure all plants were uprooted, and plants were collected in sealable bags (Johnson and Newman 2011). The plant samples were brought to the lab, sorted by taxa, and any attached root material was discarded. Samples were then dried in an oven at 105^{o}C for at least 48 hours, and weighed. Mean dry biomass (by each taxa and total native and exotic species) was calculated for the littoral zone by converting the dry biomass to grams/meter^2 and taking an average of all samples collected from the littoral zone regardless of whether they had plants.

Assessment of Curlyleaf Turions

To better understand the recruitment of curlyleaf pondweed and better predict the potential abundance of curlyleaf for the upcoming spring, turion surveys were conducted each fall. To assess curlyleaf pondweed turions in the sediment, forty sampling sites in the littoral zone (≤ 4.6m depth) were randomly selected from the full set of point intercept sampling points. The coordinates were entered into a GPS, and a boat was navigated to each point. At each point a petite ponar (225 cm^2 basal area, sample depth

~10 cm) was used to take a sediment sample. Sampling depth and substrate type were recorded. The sediment sample was then passed through a 1mm mesh sieve to remove fine sediment. The remaining sample was returned to the lab and turions were enumerated. The turions that had sprouted in the field (plants or sprouts collected with turions attached) were counted then discarded. The remaining turions were stored in transparent freezer bags and placed in a dark refrigerator at $5^{o}C$. Every 7 to 10 days the samples were examined for sprouting, and sprouted turions were counted and removed. After several weeks, when the rate of cold sprouting turions had declined, the samples were placed at room temperature ($21^{o}C$) under natural spectrum lighting for 12 hours per day. Samples were examined every 7-10 days and sprouted turions were removed and recorded. Turion viability (proportion) was calculated as the ratio of the number of sprouted turions per site (including the turions that were sprouted when collected) to the total number of turions collected per site. The total number of turions collected at each site and number of viable (sprouted) turions was expressed as number of turions per square meter and averaged across all sites.

Assessment of watermilfoil herbivores

To gain further insight on the role milfoil herbivores have on the Eurasian milfoil population in Lake Susan, repeated herbivore surveys were conducted, focusing on the milfoil weevil. The milfoil weevil, *Euhrychiopsis lecontei*, is a native weevil found in many lakes in North America. Much of the milfoil weevil's life cycle is dependent on the milfoil plant. Evidence suggests the milfoil weevil can be effective in controlling invasive populations of Eurasian watermilfoil in some lakes (Newman 2004). The milfoil weevil was found to be present in the lake in 2009. Surveys were conducted every three to four weeks to quantify the milfoil weevil population throughout the summers of 2009-2012 following sampling protocols outlined in Newman and Inglis (2009). Milfoil herbivores were sampled along predetermined transects perpendicular to the shoreline and spread geographically around the lake. Three sampling points were established on each transect, one at shallow depth (<0.75m), one at intermediate depth (0.75 to 1.5m), and one at deeper depth(>1.5m). At each sampling point, the top 0.5m of eight stems of Eurasian watermilfoil were collected and placed in a sealable bag with water. At the lab, each sample was examined with a 3x magnifying lens, plant meristems were counted, and all weevil life stages (eggs, pupae, larvae, and adults) were counted and preserved in 80% ethanol. Other milfoil herbivores, such as lepidopteran larvae, were also counted and preserved. Milfoil herbivore abundance (count/site) was compared to Eurasian watermilfoil plant frequency and biomass to determine any correlation.

Assessment of water quality

Periodically throughout the summer, several indicators of water quality were measured at 0.5m depth intervals in the water column at the deepest part of the lake. Water temperature (^{o}C) and dissolved oxygen (mg/l) were measured with a YSI 50B electronic meter. Light, photo-synthetically active radiation (PAR; quanta), was measured with a Li-cor digital meter. The depth at which surface PAR dropped to 5% was used when comparing light levels between years. Secchi depths were recorded to the nearest 0.1m.

Statistical analysis

All statistical analyses were completed using R statistical software version 2.13.1 (The R Foundation for Statistical Computing, 2011). Statistical significance was determined if p values were < 0.05. To test for statistical differences in each parameter, an analysis of variance (ANOVA) was calculated. Then a Tukey's honestly significant difference (HSD) test was performed to determine the years for which a significant difference existed. To test for differences in taxa frequency of occurrence a chi squared test was done comparing to the previous years frequency. To test for differences in qualitative abundance the ANOVA was done using mean rake rating. To test for differences between years for dry plant biomass, an ANOVA was done using the mean plant biomass by taxa. To test for differences of turion density between years, an ANOVA was done using the mean number of turions at each site. Differences in weevil populations between years were compared with an ANOVA using the mean number of weevils (at all life stages) per stem at each site

Results

Aquatic Vegetation Community

Species richness and overall abundance of submersed aquatic macrophytes in Lake Susan increased after the reduction of common carp abundance. Lake Susan had low plant richness with 13 species (submersed, floating, and emergent) documented in 2009 that increased naturally to 14 taxa by 2012 (Table 2). One species, horned pondweed (*Zannichellia palustris*) was only noted in 2009, and two species, white water buttercup (*Ranunculus longirostris*) and star duckweed (*Lemna trisulca*), recruited naturally in 2011. These two naturally recruiting taxa had very low frequencies and density (Table 2). There was generally low species richness at each survey point, with most sites having zero or one species per site. The number of sites with moderate richness (5+ taxa per site) increased from eight sites on 2009 to twelve sites in 2012. The maximum depth of plant growth (typically coontail) largely remained unchanged at approximately 4.0m during the four years.

The total dry shoot biomass of both native and invasive taxa increased significantly ($p < 0.001$) from 2009 to 2011 (Figure 1). There was an increase in the spring frequency of occurrence of most taxa from 2009 -2011 (Figure 2). Several taxa also increased in the summer frequency of occurrence (Figure 3). However, there was a decrease in overall dry shoot biomass, frequency of occurrence, and mean rake rating in 2012 (Figure 2, Figure 3, Table 3) which was likely due to low water clarity.

One of the most dramatic increases in plant abundance was for the native Canada waterweed (*Elodea canadensis*). The frequency of occurrence significantly increased each spring from 2% of sites in 2009, to 4% in 2010, 27% in 2011, and 19% in 2012 (Figure 2). Both frequency and biomass (Figure 4) of Canada waterweed had significant increases ($p< 0.001$) between 2010, 2011, and 2012.

Leafy pondweed (*Potamogeton pusillus*), a native taxa, increased in frequency from 17% of sites in 2009, to 33% in 2010, 35% in 2011, and 22% in 2012 (Figure 2). The change in frequency was significant between spring 2009-2010, summer 2010-2011, and spring 2011-2012, while the rake rating was significant ($p< 0.001$) in 2010, 2011, and 2012. Dry shoot biomass increase was only significant ($p< 0.05$) between 2009 and 2010 (Figure 4).

Sago pondweed (*Stuckenia pectinata),* another native taxa, had a significantly increase of frequency of occurrence between 2009 and 2010 and 2011-2012 (Figure 2). There was significance increase ($p<0.05$) in rake rating between 2009-2010. There was no significant change in dry shoot biomass of sago pondweed between years (Figure 4), which averaged 0.56 ± 0.53 g/m^2.

The frequency of occurrence of coontail (*Ceratophyllum demersum*) was 43% in 2009 and largely remained unchanged, with the exception of a significant increase ($p<0.05$) to 53% in 2011. Mean rake rating increased between 2009, 2010, and 2011. Biomass of coontail also increased significantly in 2010 (Figure 4). However in 2012 both frequency of occurrence and mean rake rating decreased significantly (Figure 2 and Figure 4).

Floating leaf species had high variability in frequency of occurrence. There were no significant differences in rake rating, likely due to high sampling variability. Water lotus (*Nelumbo lutea*) increased from 7% in 2009 to 11% in 2012. Yellow water-lily (*Nuphar variegata*) increased from 5% in 2009, to 8% in 2012. White water-lily (*Nymphaea odorata*) sampling was variable but averaged about 7% between 2009 and 2012 (Figure 2). Dry plant biomass values, also having high variations, showed no statistical trends (Figure 4).

The spring frequency of occurrence of the non-native Eurasian watermilfoil, decreased over the four years (Figure 2), this decrease was only significant between 2009 and 2010. The decrease in rake rating was significant ($p < 0.001$) between 2009 and other years. Biomass remained consistently low but did not differ between years (Figure 4).

Curlyleaf pondweed, an invasive exotic species, increased in spring frequency of occurrence from 17% in 2009, to 28% in 2010, and to 41% in 2011, however remained unchanged at 40% in 2012 (Figure 2). The rake rating also increased significantly ($p < 0.05$) between 2009, 2010 and 2011. However, the increase in dry plant biomass was not statistically significant ($p = 0.092$) (Figure 4). Curlyleaf pondweed showed an increase in density rating and distribution, becoming the most dominant taxa in the 1.5 to 2.5m depth range by 2010. There was also significant increase in curlyleaf turion density in the sediment (Figure 5) between 2010 and 2012. The mean turion density of the 40 randomly selected sampling points increased from 24±14 turions/m^2 in 2010 to 87±41 turions/m^2 in

2012. Because many of the sampling locations missed areas known to have curlyleaf, 12 more sampling locations were selected within high density curlyleaf stands. Of these non-randomly selected locations, the mean density increased from 148±80 turions/m^2 in 2011 to 417±223 turions/m^2 in 2012.

Milfoil Weevils

The overall density of milfoil weevils remained fairly high (Table 4) with a summer average of 0.37 ±0.15 weevils/stem in 2010, 0.16 ±0.02 weevils/stem in 2011 and 0.31 ±0.10 weevils/stem in 2012. There was a significant decrease ($p<0.001$) in the mean weevil density per sampling location between 2010-2011, but a significant increase in weevil density between 2011-2012. From 2009 to 2012 Eurasian watermilfoil decreased each year in both frequency of occurrence (Figure 2) and shoot biomass (Figure 4) in Lake Susan. The frequency of occurrence of Eurasian watermilfoil remained very low in 2012 (17% to 1%), likely due to the effect of milfoil weevils (Figure 6). The population of Eurasian watermilfoil was so low by mid-summer each year that it was difficult finding stems to collect at many of the sampling locations.

Water Quality Parameters

Springtime Secchi depth was higher in 2009-2011 then in the preceding years before the removal of carp winter of 2009 (Figure 7). This supports the hypothesis that reducing common carp populations will lead to an increase in water clarity. However, the increased clarity did not persist for the duration of the summer. Summertime Secchi depth dropped to less than 1.0m around mid July in 2010, and early August in 2011. The Secchi depth in 2012 unexpectedly declined to less than 1m around the end of May and remained less than 1m throughout the rest of the summer. This was likely due in part to record setting high temperatures in early spring 2012. The higher than usual temperatures led to an increase in phytoplankton, which corresponds with decreased summer clarity. The poor clarity limited the light available to plants. The depth at which 5% of the surface light (PAR) was available in early June was about 4.0m in 2010 and 2011, but only 1.3m in 2012 (Table 5). Another way to interpret the decreasing clarity is by the date at which the 5% threshold dropped below 1m. The water clarity dropped below the 5% threshold in mid September in 2010, about early August in 2011, and late June in 2012 (Table 5).

Temperature and dissolved oxygen profiles indicate that Lake Susan becomes stratified during the late spring and usually remains so throughout the summer. June temperature profiles (Figure 8) show a thermocline between 1.5 and 2.5m (with the exception of the warm spring of 2012, where the thermocline was between 2.0 and 3.0m). Spring dissolved oxygen profiles show a hypoxic hypolimnion in 2010 and 2011 (about 2mg/l at lake bottom), with the exception of 2012, where the hypolimnion became anoxic at 3.5m.

The typical August temperature profiles show the thermocline remains below 3.0 to 4.0m. The dissolved oxygen profiles show the hypolimnion goes anoxic about 3.5m, while the epilimnion stays fairly well mixed (Figure 9).

Discussion

After the density of common carp in the lake was reduced, the diversity and density of aquatic macrophytes steadily increased in the study lake over the four year study period. This response was similar to what has been noted in other fish biomanipulation studies such as Bajer et. al (2009), Hanson and Butler (1994), and Anderson (1950). The positive response of aquatic macrophytes was most likely due to the decrease in uprooting by carp. Increased springtime water clarity also likely contributed to the increase in the plant community. The positive plant response is illustrated by the significant increases of taxa frequency of occurrence, rake rating and increases in dry plant biomass during the first three years after carp removal.

Natural establishment of submersed aquatic vegetation species appears to show a slow and marginal increase of plant diversity, with star duckweed (*Lemna trisulca*) and water buttercup (*Ranunculus longirostris*) being the only two new species known to recruit naturally after 2009. These two taxa were found in very low abundance. Unfortunately, I was unable to obtain pre-carp removal (2008) frequency of occurrence or biomass data. However, there is documentation of 7 taxa present in very low densities (Eurasian watermilfoil, curlyleaf pondweed, coontail, sago pondweed, white waterlily, yellow waterlily, and American lotus). Lake Starring, (sharing the same local watershed Eden Prairie, MN), contains almost 500kg/ha of common carp (Bajer and Sorensen unpublished data), and could make a reasonable proxy for Lake Susan's plant community pre-carp removal. Lake Starring was also noted to have 7 taxa present (curlyleaf pondweed, coontail, sago pondweed, white waterlily, yellow waterlily, star duckweed, and horned pondweed), with lily beds consist of relatively small well defined stands, and single stems of submersed macrophytes sparsely noted in shallow (<1.0m depths). This suggests a diminished available sediment seed bank due to years of high common carp population. However, it also suggests that six taxa (cattail (*Typha, sp.*), hardstem bulrush, (*Schoenoplectus acutus*) Canada waterweed, leafy pondweed (*Potamogeton pusillus*, horned pondweed (*Zannichellia palustris*), and lesser duckweed (*Lemna minor*)) may have recruited to Lake Susan the summer after carp removal.

Native taxa with higher tolerance to lower clarity, such as coontail, Canada waterweed, and sago pondweed showed rapid increases in density and distribution during the two years after carp population reduction. In comparing other fish biomanipulations, For Hanson et al., (2006) and Lauridsen et al., (2003), it took about two years after fish manipulations for macrophytes to show large increases in occurrence. Because of the slow natural recruitment of native taxa, other methods of vegetation reestablishment, such as transplanting, may be useful in increasing the species diversity of the littoral zone.

Both native and exotic taxa increased in total biomass and frequency of occurrence after the reduction of carp. The invasive curlyleaf pondweed showed a significant increase in density and distribution. Curlyleaf pondweed increased in density and distribution through the formation of turions, as there was an continual increase in turion density in the sediment throughout the study period. This rapid increase in exotic taxa is similar to the response noted by Lauridsen et al., (2003), where four years after a large scale fish bio-manipulation, exotic taxa (Canada waterweed in this case) became the dominant species. A high density of curlyleaf pondweed can out-compete native taxa for sunlight and nutrients by forming a dense canopy on the water surface (Bolduan 1994). In Lake Susan, curlyleaf pondweed does form surface mats over about 10% of the lake surface in early summer. If left unmanaged, the increase of curlyleaf pondweed will likely pose challenges to maintaining a diverse native plant community.

The other exotic invasive species in Lake Susan, Eurasian watermilfoil, surprisingly showed a decrease in frequency of occurrence and biomass between 2009 and 2012. There was also noted a high density and persistence of the milfoil weevil, *E. lecontei*. As pointed out in Newman (2004), abundant milfoil weevil populations can be effective in controlling Eurasian watermilfoil, which also appears to be the case in Lake Susan.

The maximum depth of rooted plant growth remained unchanged, at about 4m, during the course of study. Curlyleaf pondweed (during in the spring) was the only taxa, very sparsely, found growing beyond 3m. The maximum depth of growth is largely a function of water clarity (Canfield, Langeland, and Linda 1985). The decline in plant frequency and biomass in 2012 also correlated with prolonged low Secchi depths following unusually high springtime temperatures. The decrease in water clarity in 2012 is likely a response to the increase in phytoplankton. The increase in phytoplankton and formation of an anoxic hypolimnion during the mid to late summer implies that internal phosphorous loading is likely playing a role in continued eutrophication during the summer.

While there was an increase in springtime water clarity (excluding 2012), sustained water clarity improvement throughout the summer is likely needed to increase the maximum depth of native submersed macrophyte growth and species richness.

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Tables Chapter II

Table 1. Water quality indicators for Lake Susan calculated from MPCA water quality data. Abbreviations are: Chl a-chlorophyl a; TKN-total Kjeldahl nitrogen; TP-total phosphorous. Spring defined as April thru June, Summer defined as July 1st through September. Data derived from; http://cf.pca.state.mn.us/water/watershedweb/datasearch/waterUnit.cfm?WID=10-0013-00##

		Secchi (m)	
	Spring	Summer	Mean
2008	2.2 ±0.4	0.9 ±0.2	1.1±0.2
2009	2.5 ±0.2	0.7 ±0.1	1.4 ±0.2
2010	2.0 ±0.2	1.1 ±0.1	1.6 ±0.2
2011	3.3 ±0.5	1.0 ±0.2	1.7 ±0.4
		TKN (mg/L)	
	Spring	Summer	Mean
2008	11.3 ±9.4	2.7 ±0.2	6.0 ±3.6
2009	1.5 ±0.1	1.9 ±0.2	1.8 ±0.1
2010	1.0 ±0.1	1.5 ±0.1	1.2 ±0.1
2011	1.0 ±0.1	1.5 ±0.1	1.4 ±0.1
		Chl a (ug/L)	
	Spring	Summer	Mean
2008	9.9 ±3.3	66.1 ±12.1	52.0 ±11.6
2009	12.3 ±3.5	55.1 ±4.6	37.9 ±4.9
2010	9.5 ±2.8	39.3 ±6.9	23.6 ±4.9
2011	4.6±1.6	36.6±5.1	26.7±5.5
		T P (mg/L)	
	Spring	Summer	Mean
2008	0.98 ±0.936	0.10 ±0.017	0.37 ±0.285
2009	0.04 ±0.005	0.11 ±0.013	0.08 ±0.011
2010	0.05 ±0.006	0.08 ±0.011	0.06 ±0.007
2011	0.02 ±0.002	0.08 ±0.008	0.06 ±0.009

Table 2. List of species found in Lake Susan during all surveys. Highest frequency of occurrence each year listed. P= present but not found in sample location, T= transplanted but not found in sample location, Tx.xx= transplanted and frequency found in lake-wide survey (found in a nearby sampling location).

Common Name	Scientific Name	Code	2009	2010	2011	2012
Emergent species						
Cattail	*Typha spp.*	Typh	P	P	P	P
Hardstem bulrush	*Schoenoplectus acutus*	Sacu	P	P	P	P
Submerged species						
Coontail	*Ceratophyllum demersum*	Cdem	0.43	0.37	0.53	0.34
Muskgrass	*Chara sp.*	Chara	T	T	T	T
Canada waterweed	*Elodea canadensis*	Ecan	0.02	0.04	0.27	0.19
Northern watermilfoil	*Myriophyllum sibiricum*	Msib	T	T	T	T
Eurasian watermilfoil	*Myriophyllum spicatum*	Mspi	0.35	0.21	0.14	0.17
Bushy pondweed	*Najas flexilis*	Nfle	T	T 0.08	T 0.05	T
Curlyleaf pondweed	*Potamogeton crispus*	Pcri	0.17	0.28	0.41	0.40
Longleaf pondweed	*Potamogeton nodosus*	Pnod	0	T	T	T
Narrow leaf pondweed	*Potamogeton pusillus*	Ppus	0.17	0.33	0.33	0.22
Flatstem pondweed	*Potamogeton zosteriformis*	Pzos	0	T	T	T
Water crowfoot	*Ranunculus longirostris*	Rlon	0	0	P	P
Sago pondweed	*Stuckenia pectinata*	Spec	0.03	0.10	0.07	0.08
Wild celery	*Vallisnaria americana*	Vame	T	T	T	T
Horned pondweed	*Zannichellia palustris*	Zpal	0.06	0	0	0
Water stargrass	*Zosterella dubia*	Zdub	T	T	T	T
Floating-leaf Species						
Star duckweed	*Lemna trisulca*	Ltri	0	0	0.01	P
Lesser duckweed	*Lemna minor*	Lmin	0.03	P	0.05	0.01
Water Lotus	*Nelumbo lutea*	Llut	0.07	0.08	0.11	0.11
Yellow lily	*Nuphar variegata*	Nvar	0.05	0.09	0.06	0.08
White lily	*Nymphaea odorata*	Nodo	0.10	0.02	0.08	0.06
Total Richness (# transplanted)			**18(5)**	**19(7)**	**21 (7)**	**21(7)**

Table 3. Mean rake rating (0-5, with 5 being most dense) by taxa and by year from the point intercept surveys.

	Spring				Summer			
	2009	2010	2011	2012	2009	2010	2011	2012
Whole Rake	0.86	1.21	1.58	1.46	0.78	2.07	1.55	1.25
Coontail	0.52	0.60	0.88	0.46	0.60	1.24	0.98	0.66
Canada waterweed	0.00	0.03	0.34	0.23	0.01	0.02	0.45	0.36
Narrowleaf pondweed	0.15	0.62	0.58	0.36	0.13	0.63	0.73	0.42
Water Lotus	0.00	0.07	0.06	0.08	0.05	0.15	0.18	0.31
Yellow Lily	0.05	0.10	0.18	0.10	0.03	0.24	0.14	0.19
White Lily	0.09	0.07	0.08	0.01	0.06	0.17	0.17	0.12
Star Duckweed	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.00
Sago pondweed	0.01	0.06	0.08	0.01	0.03	0.24	0.08	0.11
Curlyleaf pondweed	0.16	0.37	0.81	0.95	0.05	0.00	0.08	0.45
Eurasian watermilfoil	0.38	0.18	0.18	0.32	0.29	0.12	0.11	0.17

Table 4. Mean number of milfoil weevils (of all life stages) per stem from all surveys by year.

	2010			2011			2012	
Date	Weevils/ stem	1SE	Date	Weevils/ stem	1SE	Date	Weevils/ stem	1SE
6/4	0.82	0.11	6/7	0.21	0.06	6/1	0.47	0.11
6/17	0.87	0.14						
7/6	0.27	0.04	7/7	0.16	0.04	6/28	0.12	0.08
7/27	0.19	0.05						
8/17	0.04	0.02	8/3	0.15	0.03	8/8	0.33	0.33
9/4	0.04	0.02	9/1	0.13	0.02			
mean	0.37		mean	0.16		mean	0.31	
1SE	0.15		1SE	0.02		1SE	0.10	

Table 5. Depth of 5% of surface light by date for Lake Susan.

Date	Depth of 5% Surface Light
5/14/2010	4.5
6/2/2010	4.0
6/17/2010	3.0
6/30/2010	2.0
7/6/2010	2.0
7/21/2010	1.8
9/17/2010	1.0
6/7/2011	4.0
6/14/2011	3.5
6/27/2011	3.0
7/7/2012	3.0
8/4/2011	1.5
8/10/2011	1.0
8/31/2011	0.5
6/7/2012	1.3
6/13/2012	1.2
6/28/2012	1.0
7/3/2012	1.3
7/12/2012	1.0
8/6/2012	0.7
10/10/2012	1.3

Figures Chapter II

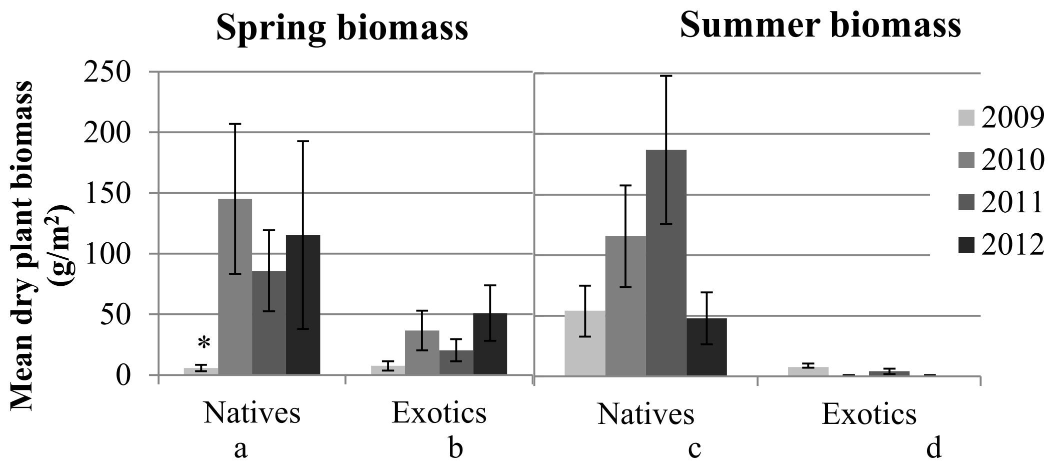

Figure 1. Mean dry shoot biomass of natives and exotics (*M. spicatum* and *P. crispus*) in Lake Susan in 2009, 2010, 2011, and 2012. Error bars indicate one standard error, letters indicate grouped ANOVA test, * indicate a p value <0.5.

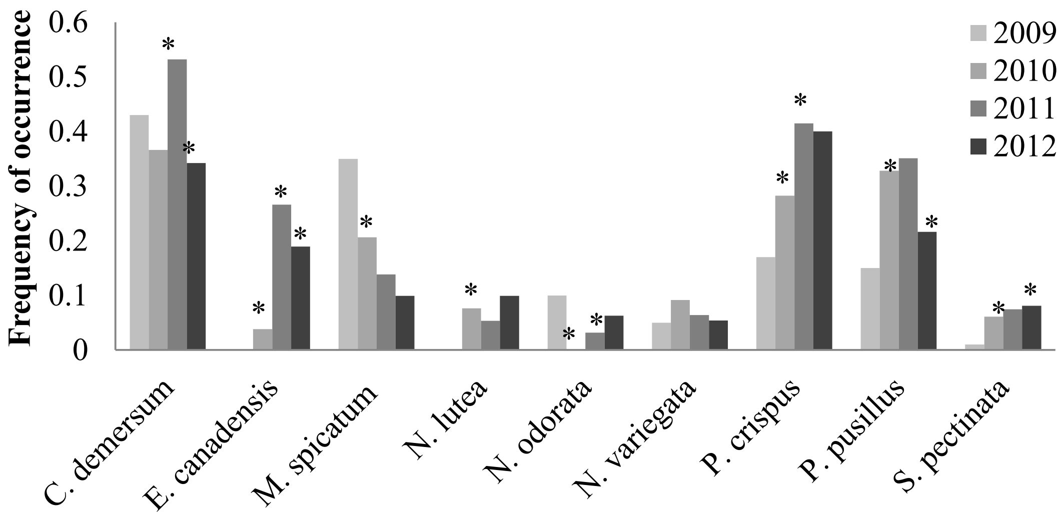

Figure 2. The frequency of occurrence of the most common taxa found in Lake Susan during the spring point intercept surveys in 2009, 2010, 2011 and 2012. Not listed are taxa that had very low frequency of occurrences. Significant difference (p < 0.5) to previous year indicated by *. See Table 2 for species common names.

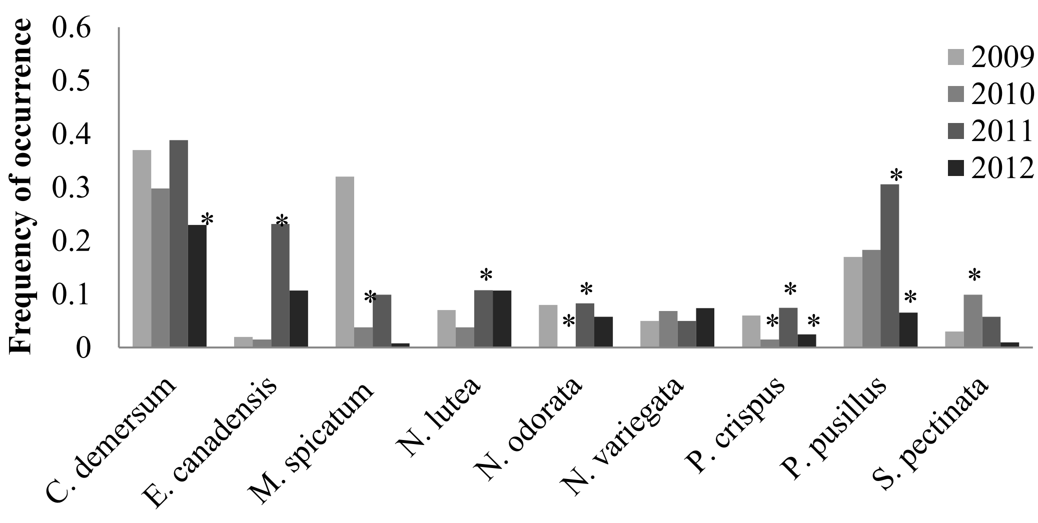

Figure 3. The frequency of occurrence of the most common taxa found in Lake Susan during the summer point intercept surveys in 2009, 2010, 2011, 2012. Not listed are taxa that had very low frequency of occurrences. Significant difference (p < 0.5) to previous year indicated by *. See Table 2 for species common names.

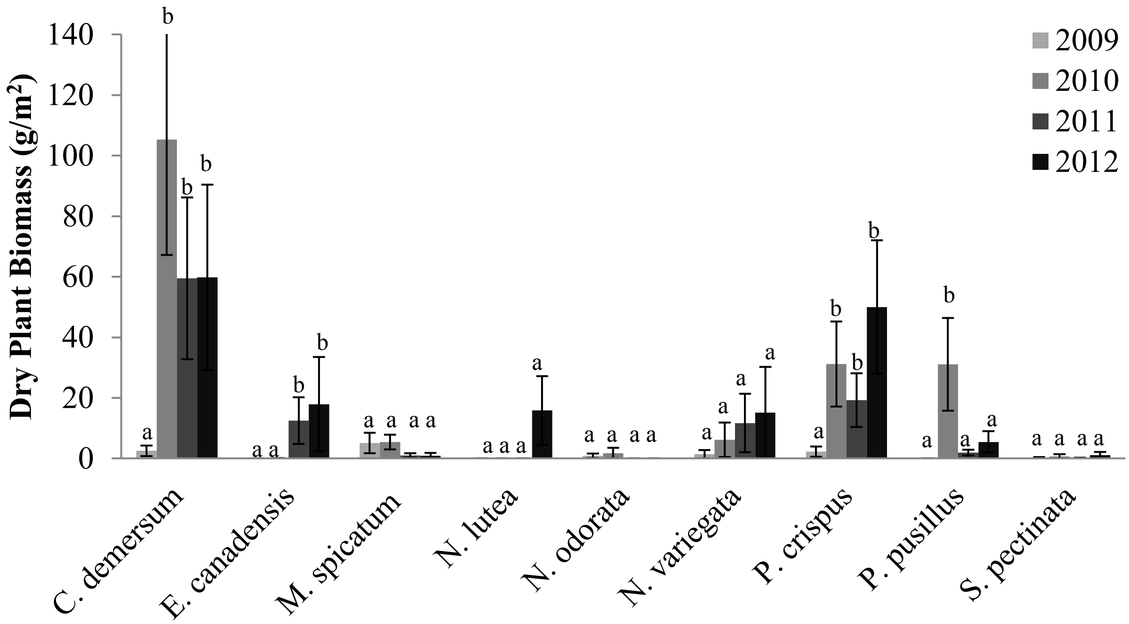

Figure 4. Mean dry plant shoot biomass (g/m^2) of common species for Lake Susan during spring point intercept surveys. Error bars indicate one standard error. Letters indicate significant differences between years p < 0.5.

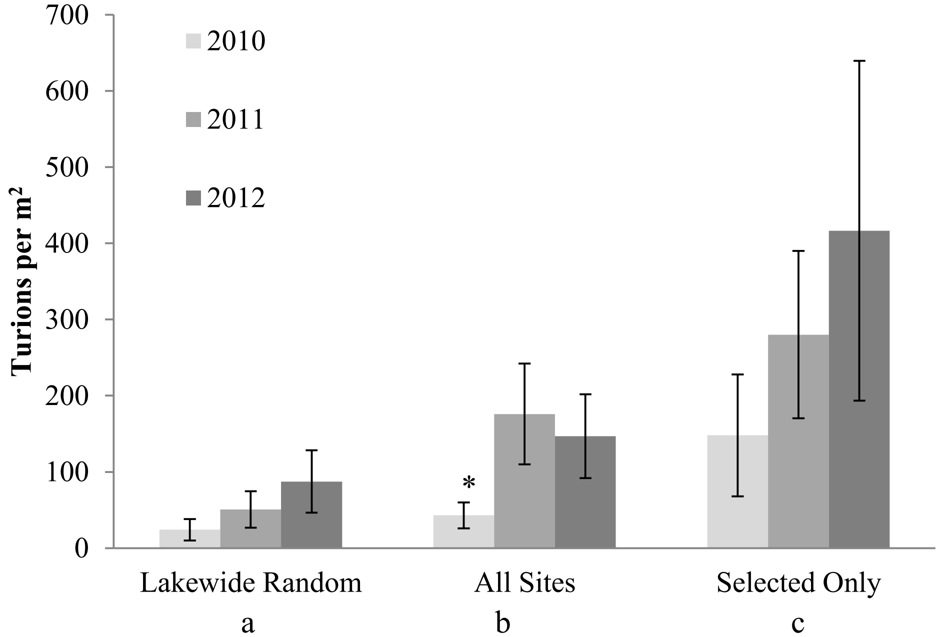

Figure 5. Mean density of curlyleaf turions (per m^2) found in the sediment during October sampling in 2010, 2011, and 2012. Values shown are from the 40 randomly selected lake-wide sampling points; all points sampled; and 12 selected high biomass, sampling points. Error bars indicate one standard error, letters indicate grouped ANOVA test, * indicates p value <0.5.

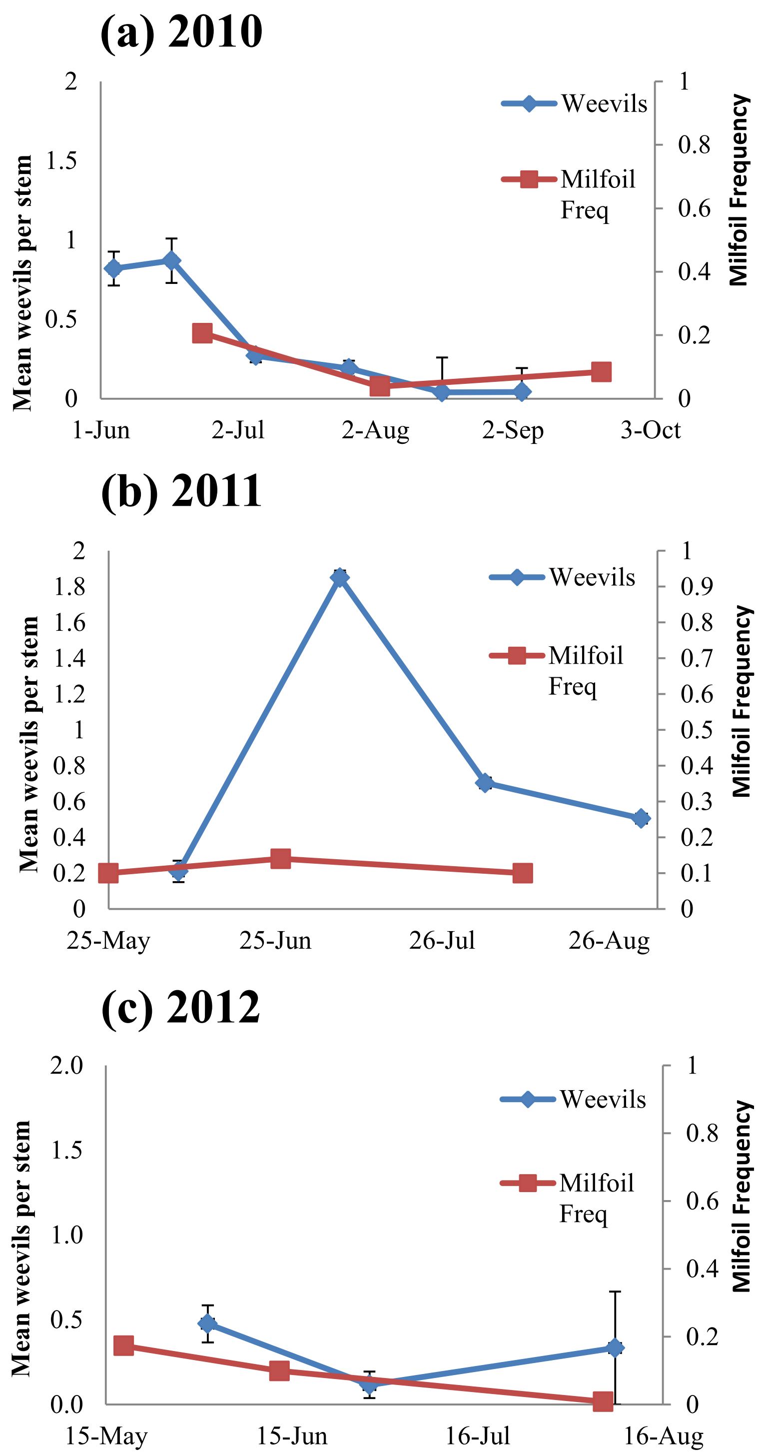

Figure 6. Mean number of milfoil weevils (all life stages) per stem (blue) and lake-wide frequency of occurrence of Eurasian watermilfoil (red) during summer sampling in 2010 (a), 2011 (b) and 2012 (c).

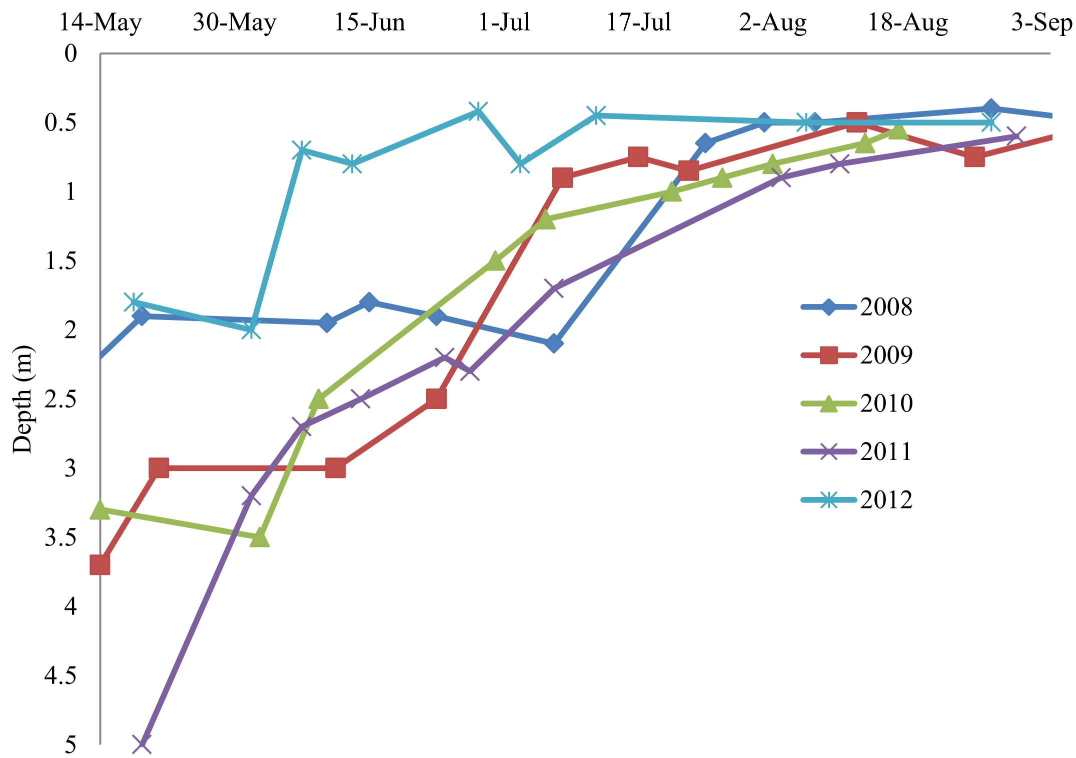

Figure 7. Summer Secchi depths in Lake Susan during 2008 thru 2012 sampling. 2008 and 2009 data collected by P. Bajer and P. Sorenson.

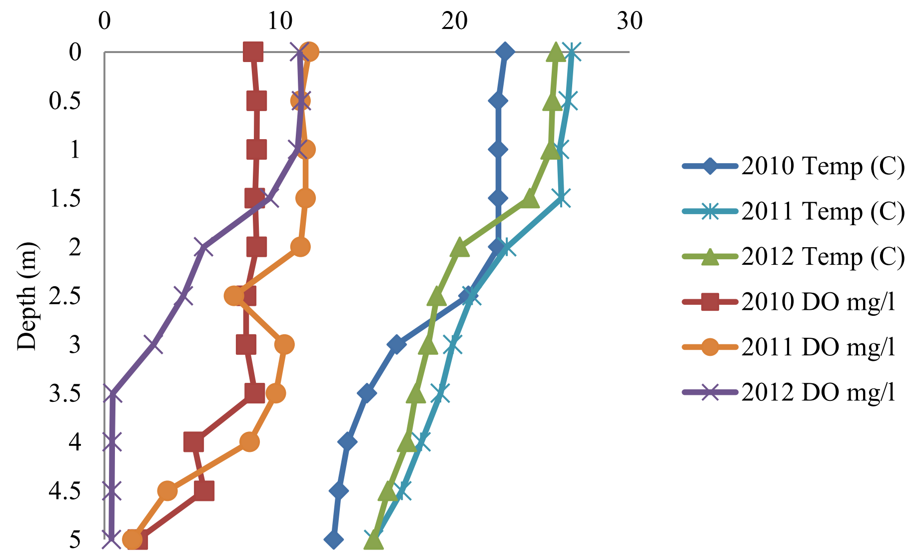

Figure 8. Typical June dissolved oxygen (mg/l) and temperature (^oC) profiles in Lake Susan. Profiles taken 2 June 2010, 7 June 2011, and 7 June 2012.

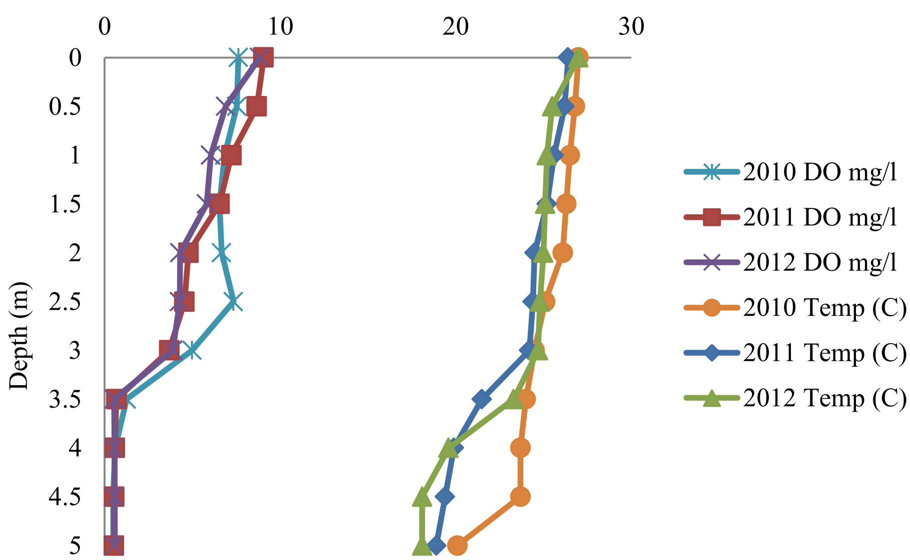

Figure 9. Typical summer dissolved oxygen (mg/l) and temperature (^oC) profiles in Lake Susan. Profiles taken on 13 August 2010, 10 August 2011, and 6 August 2012.

Chapter III

Transplanting Aquatic Macrophytes to Restore the Littoral Submersed Plant Community After Carp Removal

Summary

To promote the growth and expansion of native submersed macrophytes after the removal of carp from Lake Susan (Carver County, Minnesota), six native taxa were transplanted from nearby Lake Ann into Lake Susan. The six species were muskgrass (*Chara spp.*), wild celery (*Vallisnaria americana*), northern watermilfoil (*Myriophyllum sibiricum*), bushy pondweed (*Najas flexilis*), water-stargrass (*Zosterella dubia*) and flat-stem pondweed, (*Potamogeton zosteriformis*). In a series of four separate experiments, a total of sixteen shallow (0.5-1.0m) and ten deep (1.0-1.5m) plots, each containing five taxa, were transplanted along undeveloped locations around the lake. The survival and expansion of plants were measured and compared against several environmental factors, such as planting depth, relationship to stands of lily beds, timing of transplanting, sediment texture, soil organic matter and available nitrogen. Transplanting whole plants in shallow water (<1 meter) was generally successful over the four year duration of this study. Most species transplanted survived during the initial growing season, however overwintering success was a better predictor of long-term success than was initial survival. Water-stargrass had high long term survival and large expansion making it the most successful taxa. Although wild celery had high a survival rate, it showed very slow expansion. Bushy pondweed had variable survival but when it survived it generally expanded well, whereas muskgrass and northern watermilfoil generally showed poor survival and expansion. The most important environmental factor in the success and expansion of transplants was light availability. Plants transplanted in deeper zones (approximately 1.4m depth) failed to establish due to low light (summer Secchi depths

were often <1.0m). Flatstem pondweed failed to establish as it was only planted in the deeper locations. Lake wide, both native and non-native aquatic macrophytes responded favorably to the removal of high carp densities. Transplanting whole submersed aquatic macrophytes can be used to help restore the littoral community.

Introduction

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health by reducing sediment re-suspension and providing habitat for aquatic organisms ranging from algae consuming zooplankton, to fish and other wildlife (Valley et al. 2004, Horppila and Nurminen 2003, Sondergaard et al. 2003). A healthy aquatic macrophyte community consists of a high diversity of native taxa (Madsen 1997). High densities of invasive common carp uproot aquatic macrophytes and stir up sediments, encouraging a turbid water state with a low density of macrophytes (Schrage and Downing 2004, Bajer et al, 2009). Reducing the population of the benthic feeding common carp can increase the distribution of the aquatic plant community. However it can take several years for plants to return (Hanson and Butler 1994), and this has led to experimentation with transplanting adult aquatic plants as a means for plant community restoration.

The distribution of aquatic macrophytes, and subsequent success of transplanting them, is related to a number of environmental factors including light, temperature, sediment structure and fertility. One of the most important environmental factors for the growth of submersed macrophytes is light availability (Scheffer 1998). The depth at which light is available to plants is a function of water clarity, where clearer water allows plants to occupy greater depth zones, and thus have a greater distribution around the lake. Poor water clarity restricts aquatic vegetation to shallow depths. Temperature was found to be highly influential to total plant biomass production, with an increase in temperature relating to an increase in biomass up to a critical temperature (Barko and Smart 1981). The growth of submersed aquatic plants, like that of terrestrial plants, is in part influenced by the composition and fertility of the substrate in which they grow (Sculthorpe 1967). Fertility of the littoral sediment can be influenced by wave energy reducing accumulation of fine sediments and decreasing the growth of submersed vegetation (Doyle 2001). Although it has been shown that protection from wave energy can be important (Smart et al. 1996) for transplanting aquatic macrophytes, it has not been demonstrated that natural barriers such as lily beds can be a means for protection of transplants. Timing of transplants was found to be one of the most important variables that lead to successful establishment of the emergent macrophytes, where early summer plantings resulted in higher survival rates than late summer or early fall plantings (Vanderbosch and Galatowitsch 2011).

Once high populations of benthivorous fish, such as common carp, are removed from the lake, sediment re-suspension should decrease, water clarity should improve, and macrophytes should increase (Bajer et al. 2009). However, it can take several years for aquatic macrophytes to establish in high densities after fish removal, and often the taxa that do establish are non-native invasives (Hanson and Butler 1994, Hanson et al. 2006, Lauridsen et al. 1994). Manipulation of the littoral community may be necessary to promote the establishment of native vegetation rather than invasive taxa.

Transplanting can be an effective method for introducing submersed aquatic macrophytes into water bodies where few propagules exist (Smart et al. 1998, Lauridsen et al. 2003, Smiley and Dibble 2006). Although transplanting submersed aquatic macrophytes has been shown to be successful, the majority of the case studies were done in reservoirs in southern latitudes of the Unites States or in more temperate climates of Europe (Dick et al. 2004a, Dick et al. 2004b, Flemming et al. 2011, Lauridsen et al. 2003). Quantifying the growth and expansion rate of the transplants, along with environmental variables related to submerged macrophyte growth such as depth, light intensity, sediment texture, timing of transplanting, soil organic matter and available nitrogen, may help further narrow the factors that lead to successful future transplanting. Although there have been several studies on the methodology and successfulness of transplanting submersed macrophytes as a means for plant community restoration, there has been little done to document the rate of expansion within the lake. Documenting the success and expansion rate of transplanted submersed aquatic macrophytes will assist with planning and development of future littoral vegetation habitat restoration projects.

The primary goal of this project was to evaluate the reintroduction of several native submersed macrophyte taxa into a lake after the carp population was reduced. I examined environmental variables that pertain to vegetation growth, and compared them to the survival and colonization rate of the transplanted species. To assess the effectiveness of transplanting macrophytes, four separate experiments were conducted.

The aim of experiment one was to assess the initial and long term survival of five transplanted taxa and compare the difference in growth (height) between caged locations protected from large herbivore access, and locations open to herbivore access. The aim of experiment two was to increase the number of transplant locations geographically around the littoral zone, and determine if protection by floating leaf lily beds enhanced growth. Experiment three tested transplant survival and expansion in deeper water (approximately 1.4m), and experiment four evaluated whether earlier planting of transplants would improve survival of deeper transplants.

A littoral zone consisting of a diverse community of native aquatic macrophytes provides numerous important ecological functions for lake health. Restoring the natural aquatic macrophyte community will help create a more resilient ecosystem and help reduce the potential for invasive macrophytes species to establish in high densities. It has been shown to take many years for macrophytes to re-establish on their own after a reduction on carp populations. Transplanting submersed aquatic macrophytes can be a means of restoring the littoral plant community.

Methods

Lake Susan, Carver County (Minnesota Department of Natural Resources (MN DNR) Division of Waters (DOW) ID 10-001300), is a eutrophic lake in Central, Minnesota USA. The lake covers about 38 hectares, with approximately 30 hectares less than 4.6m in depth (the MN DNR defined littoral zone) and a maximum depth of 5.2m (MN DNR Lake finder 2011). Prior to carp removal, Lake Susan had low summer Secchi depths, few submerged aquatic plants, and a high abundance of common carp (Bajer and Sorensen 2012). Using under ice seining in the winter of 2009, approximately 78% of the carp were removed from the lake (carp standing stock reduced from 307kg/ha to 65kg/ha) in an attempt to determine if removal of carp would enhance water clarity and improve the native plant community (Bajer and Sorensen 2012).

It has been found to take several years for aquatic macrophytes to naturally recruit. It was suspected there were few viable propagules in the benthos due to high carp abundance and by observations made in 2008. Six taxa of native species were transplanted from nearby Lake Ann, Carver County, MN (DOW ID 10-001200) into Lake Susan, to promote the growth and expansion of native macrophytes after the removal of carp. The source lake (Lake Ann) is located about 3km upstream of Lake Susan. It is within the same watershed and has good water clarity (summer Secchi depths of 2.5 to 4m) and high species richness (25 submersed, emergent and floating leaf species).

Several criteria were considered for selecting the species to transplant: (1) all species were native to Minnesota; (2) species were expected to have high likelihood of success as determined by Smart et al., (1996); (3) species selected were in adequate abundance in the source lake, and (4) species were not already found in Lake Susan. The six species were muskgrass (*Chara spp.*)*,* wild celery (*Vallisnaria americana*), northern watermilfoil (*Myriophyllum sibiricum*), bushy pondweed (*Najas flexilis*), water-stargrass (*Zosterella dubia*) (shallow only) and flat-stem pondweed*,* (*Potamogeton zosteriformis*) (deep only). Aside from the above mentioned reasons, there were a few specific justifications used in species selection. Wild celery was selected for its benefit for waterfowl (Sponberg and Lodge 2005). Northern watermilfoil was selected in part due to its ability to host the native milfoil weevil, *Euhrychiopsis lecontei,* a potential biocontrol agent for invasive Eurasian watermilfoil (*M. spicatum*) (Newman 2004). The milfoil weevil was present in Lake Susan, and is also hosted by Eurasian watermilfoil. The establishment of northern watermilfoil could assist with stabilizing future milfoil weevil populations in the lake. Muskgrass was chosen due to many favorable characteristics, including direct uptake of nutrients from the water column, and tendency to carpet the sediment (Kufel and Kufel 2002) providing sediment stability. Because water-stargrass was found only in the shallow zone (<0.7m) of the source lake, it was thought to be a poor candidate for deeper transplanting. Instead, the deeper growing flat-stem pondweed, was selected for transplanting to the deeper plots.

Care was used to ensure only the preselected species were collected. Although the source lake (Lake Ann) contained two invasive exotic species, Eurasian watermilfoil and curlyleaf pondweed (*Potamogeton crispus*), both species were also present (at low densities) in Lake Susan, negating the concern for inadvertent introduction. Transplants were collected from the source lake by snorkeling and wading in 0.5m to 1.25m depths. A small garden spade was used to dig up the sediment around the roots of mature plants, and care was taken to not damage root structures. The roots were gently rinsed of sediment and whole plants were placed in a large cooler with like taxa and lake water. The coolers of transplants were stored at room temperature overnight, and transplanted to Lake Susan the next day.

Transplant locations were noted as plots during the study. Each plot was located away from developed shoreline areas, as to not interfere with recreational boating (Figure 1). The location of each plot was recorded by GPS. Each plot contained five transplant sites, spaced 1m or 2m apart. Each site was marked with a small labeled PVC pipe pushed into the substrate to aid in locating sites for future monitoring. One species was randomly selected to be transplanted at each site. The plants were transplanted by digging a small hole (approximately 5cm) either by hand (in softer substrates) or with a small hand trowel (harder substrates). Bare roots were placed in the small hole and a steel sod staple (10cm) was used to hold the roots in place. The roots, together with the sod staple, were then covered with sediment. Muskgrass, which has no roots, was transplanted as a cluster approximately $500cm^3$ held to the substrate with one sod staple.

Sediment Collection

Sediment bulk density, organic matter, pore water nitrogen and total exchangeable nitrogen were measured at each plot. Sediment core samples were taken at each end of the transplant plots, resulting in two samples per plot (approx. 8m apart). The core samples were collected with a 5cm diameter PVC pipe forced about 0.25m into the sediment. A one-way check valve on top of the pipe helped to retain the sediment in the core tube upon retraction from the sediment. The top 10cm of sediment from the core tube was then collected by using a plunger to force the sediment through the top and using a large spoon to scrape the sediment into a sealable bag. At each sampling location the associated transplanted taxa was noted. Bulk density (g/ml) was calculated by weighing a 10ml subsample of homogenized sediment in a pre-weighed crucible (A), drying the sample at 105°C for 48 hrs and obtaining weight (B) after cooling. Bulk density was calculated by: $Bulk\rho = (\frac{(B-A)}{10ml})$. The percent organic matter of the sample was calculated by combusting the sample for 4 hours at 550°C , cooling and re-weighing (C). Percent organic matter was calculated as $\%OM = \left(\frac{B-C}{B-A}\right) * 100$. Pore water ammonia and soil bound nitrogen were measured to evaluate the total amount of nitrogen available. Pore water was collected by spinning a 50ml subsample of homogenized sediment for 15 minutes at 3000 rpm in a centrifuge and decanting the supernatant. A Hach pH meter with ammonia electrode (Orion model 9512) was used to measure ammonia in the supernatant following procedures described by Frazier (1996). The available sediment-bound nitrogen was extracted by adding 50ml of 2 molar KCL solution to the 50ml sediment subsample remaining after pore water was extracted. The sample was homogenized for one hour and filtered through (Whatman 1) 415 qualitative filter paper. The extracted nitrogen from the filtrate was then measured using the previously mentioned ammonia probe and methods given above. Total nitrogen was calculated by adding the value from pore water with the value from the KCL extraction.

Assessment of water quality

Periodically throughout the summer, several indicators of water quality were measured at 0.5m depth intervals in the water column at the deepest part of the lake. Water temperature (°C) and dissolved oxygen (mg/l) were measured with a YSI 50B electronic meter. Light PAR (quanta) was measured with a Li-cor digital meter. The depth at which surface PAR dropped to 5% was used to compare light levels among years. Secchi depth was recorded to the nearest 0.1m to measure water transparency.

Experiment 1: Establishing methodology, shallow plots planted in 2009

The primary goal of experiment one was to assess the survival and growth of the initial selected macrophyte species, and to compare the difference in growth between locations protected and not protected from herbivore access. Assessment of the survival and growth (height) of the transplants were the metrics used for comparison. On 20, August 2009, four plots were located along undeveloped reaches of shoreline in Lake Susan, two along the western shore and two along the eastern shore in water depths of 0.4 to 0.8m (Figure 1). One plot on the western end and one plot on the eastern end of the lake was enclosed with a 1.5m x 6m wire barrier (2.5cm mesh) to prevent herbivore access. Each plot contained five transplant sites and five control sites. At each transplant site, four stems of one of the five taxa were planted (one stem was planted about 5cm in each cardinal direction from the marker post). The five species were muskgrass*,* water-stargrass*,* northern watermilfoil*,* bushy pondweed*,* and wild celery. Plant height was measured about every three weeks during the growing seasons of 2009, 2010, 2011 and 2012 to evaluate survival and growth. The maximum growth and survival values each summer were used to normalize differences in growth phenology among species. Control sites were established to determine naturally recruiting taxa, and were located about one meter shoreward from each of the transplant sites and marked with a small PVC pipe. At each control site, one plant in each cardinal direction from the marker post (four plants per site) was identified and height measured.

Experiment 2: Lake-wide shallow plots, planted in 2010

Twelve more plots of the same five taxa were transplanted at shallow (0.5 - 0.75m depth) locations distributed around the lake on 01 August 2010. Six of the plots were planted shoreward of lily stands (protected), and six of the plots were planted open to wave action (open) to evaluate whether stands of floating leaf species (white waterlily (*Nymphea odorata*), yellow waterlily (*Nuphar variegata*), and water lotus (*Nelumbo lutea*)) provided adequate wave protection. Transplant plot locations were distributed around the undeveloped stretches of shorelines to minimize human interference and to maximize potential for lake-wide colonization. One "protected" plot and one "open" plot were located reasonably close to each other in an attempt to have similar fetch, shading and substrate characteristics. Two sediment samples per plot were taken and available nitrogen, bulk density, and percent organic matter measured and correlated to growth. Two light PAR (quanta) measurements per plot were recorded with a Li-cor digital meter on 04 August 2011 to approximate differences between plots.

The species planted were muskgrass, water-stargrass, northern watermilfoil, bushy pondweed, and wild celery. Each site had 10 stems planted within a 0.25 square meter area. Muskgrass was transplanted as 10 clusters approximately 500cm^3 each. The success of the transplanting was assessed at each site every three to four weeks during the growing season for average plant height and area of coverage. While survival of transplants is an initial indicator of success, the preferred desired outcome is an increase in coverage within the littoral zone. Expansion of the transplant sites was measured to further understand the rate transplanted taxa could expand to fill a given area within the littoral zone. Coverage was calculated by measuring area of homogenous growth (cm^2) as well as the area of presence. The homogenous area was defined as the area in which approximately ≥ 90% of the plants were made up of the transplanted taxa. Comparing the area of homogenous growth is a method to indicate the expansion of that transplanted taxa (Figure 2). A very large homogenous area would indicate stand development of the taxa. The area of presence was defined as the area in which the species was present, but not dominant (Figure 2). A very large area of presence would indicate the taxa is colonizing the littoral area but as part of a more heterogeneous plant community.

Experiment 3: Transplanting depth, deeper plots planted in 2010

Transplant success in deeper water was tested at four additional plots (two plots on the east side and two on the west side of the lake). The plants were transplanted in depths of 1.2 m to 1.6m on 22 July 2010 (Figure 1). The five species transplanted were muskgrass, flatstem pondweed, northern watermilfoil, bushy pondweed, and wild celery. Assessment and statistical analysis of the success and growth of these transplants was similar to that of experiment 2. Each site was assessed about every four weeks during the growing seasons of 2010, 2011 and 2012 for average plant height and area of coverage. Coverage was calculated by measuring area of homogenous growth (cm^2) as well as the area of presence.

Experiment 4: Earlier planting of deeper transplants 2011

Six more deeper plots (1.2m to1.6m) were transplanted on 24 June 2011 (Figure 1), using the same five taxa as experiment 3, to evaluate if earlier planting improved the success of the deeper transplants. The transplanting was timed to allow the plants to grow as much as possible in the source lake, while allowing three to four weeks to establish in Lake Susan before Secchi depth dropped below the transplanting depth of 1.4m.

Statistical Analysis

All statistical analyses were completed using R statistical software version 2.13.1 (The R Foundation for Statistical Computing, 2011). Mixed effect models were calculated using the lme4 package (Bates 2010). Statistical significance was determined if p values < 0.05.

Linear mixed-effects models test for significant fixed effects, such as growth, and develop predictive models for potential growth factors. Fixed effects are parameters associated with an entire population or from observations taken on all treatments of interest, and random effects are associated with individual experimental units drawn at random from a population.

A generalized linear mixed effects model was used to test for differences in survival between open and caged locations. The generalized linear mixed effect model summarizes survival as a binary value. A site was considered successfully established if any transplants were found to have survived at a given transplant site. Plot number was modeled as random-effects, and growth factors modeled as mixed effects.

A linear mixed effects model was used to test for differences in growth rate as related to sediment variables The general equation for the linear mixed-effects model, as noted by Laird and Ware (1982), is $y = x_i\beta + Z_i b_i + \varepsilon_i$ where y is growth area, $x_i\beta$ are the mixed effects (treatment (open vs. protected), sediment bulk density, organic matter and total nitrogen), $Z_i b_i$ are the random effects (plot number), and ε_i are the residual errors. A linear mixed effect model was used to compare the sediment characteristics to the expansion of associated transplanted taxa in the sampling locations. The mixed effect model analysis was performed twice, once with homogenous area as the random effect, and again with area of presence as the random effect. This compares the fixed effects ("Plot" and "Plant") with the mixed effects ("treatment", "density", "organic matter", and "total nitrogen").

Results

Experiment 1: Caged versus un-caged, shallow plots planted in 2009

Overall there was good plant establishment in 2009, with all taxa having at least 50% survival (Table 1). Plants that overwintered in 2010 generally survived well in the subsequent years (Figure 3). During 2009, the initial growing season, plants in caged locations survived slightly better than open locations (Figure 4), but there was no difference in growth (measured as height).

Wild celery had the highest overall survival with plants found in 81% of the sites in 2009, 98% of the sites in 2010, 90% of sites in 2011, but only in 56% of sites in 2012 (Figure 3). Water stargrass also had a high establishment, being found in 94% of the sites in 2009, 88% of sites in 2010, 81% of sites in 2011, but only 31% of sites in 2012. Overwintering success may play a role in the reduced survival of northern watermilfoil. Although the majority (75%) of the sites had successful plant establishment in 2009, only 13% of the sites had northern watermilfoil in 2010, 6% in 2011 and 4% in 2012 (Figure 3). This downward trend was also noted for bushy pondweed, which had a high initial establishment with 100% of sites surviving in 2009, but only 38% of sites in 2010, 6% in 2011, and no plants surviving in 2012. Overwintering success rate did not seem to play a critical role in the long term survival of muskgrass. Initial survival of muskgrass was good, with survival at 56% of the sites in 2009, but it only reappeared in 6% of the sites in 2010. Interestingly, 50% of the sites had muskgrass present in 2011 and 31% in 2012 (Figure 3).

The initial methods were designed to monitor the success and growth of the individual stems that were planted. Using the original methods for monitoring the transplants after the initial growing season proved to be problematic. During the subsequent growing season, if a propagule didn’t re-emerge within approximately 15cm radius from where it was originally transplanted, it wasn’t noted. This bias in sampling under represented annuals, where the next years’ propagule may emerge a meter away. For example, bushy pondweed showed a low survival being found at only 6% of the originally planted locations in 2011. However 50% of the sites showed success within a few meters of the originally planted area. To further monitor expansion, area of presence within these plots was measured at the end of the second growing season, 2011.

Water stargrass showed the greatest area of presence with all sites averaging nearly $36m^2$ in area (Table 2), and a mean area of $44.5m^2$ at only successful sites. Although wild celery had a high survival rate of 94% in 2011, its average expansion was lower with all sites averaging 16 m^2 in coverage (Table 2), and $24.0 \pm 18m^2$ at successful sites. Bushy pondweed had a 50% success rate and averaged $8.3 \pm 11.86m^2$ in area of presence (surviving sites averaged $35m^2$). Muskgrass had better survival in 2011 compared to 2010, with success rate of 56% (Figure 3). However expansion was very low, averaging only $0.28 \pm 0.25m^2$ in area of presence (surviving sites averaged $0.6 \pm 0.3m^2$).

Natural recruitment

The most frequently occurring species that established in the control cites was sago pondweed (*Stuckenia pectinata*) found in 42% of sites in 2010, narrowleaf pondweed (*P. pusillus*) in 21% of sites and coontail found in 18% of control sites. Because the expansion of water stargrass, wild celery, and bushy pondweed was greater than one meter during the 2011 growing season, the distance between the control site and treatment site, they often grew into the control sites. Thus the frequency and species composition at those sites after the first year is difficult to interpret.

Experiment 2: Lake-wide shallow plots, planted in 2010

The taxa transplanted in 2010 were successful in initial establishment. By the end of the first growing season (2010) 100% of the sites contained viable water stargrass, wild celery, and bushy pondweed plants, 92% of sites with contained viable northern milfoil plants, and 58% of sites contained muskgrass (Figure 5). Beyond the initial establishment, most plants survived overwinter into 2011 and 2012 (Figure 6). There was no significant difference in survival rates between open vs. protected areas in 2011. However the survival of all transplants decreased in 2012, along with the lake-wide plant community, due to decreased water clarity.

While survival during the first growing season (2010) was high for all taxa, expansion was not. No sites showed an increase in either homogenous area or area of presence from the original planting of $0.25m^2$ in the initial growing season (Table 3). There was also little difference in homogenous area between open and protected sites in 2010 and 2011. In 2012, there was a significant increase in protected sites containing bushy pondweed and water stargrass in both homogenous area (Figure 6) and area of presence (Figure 7). However, the opposite was true for northern watermilfoil where plants at all protected sites failed to survive.

Summarizing the survival rate of each taxa by their area of presence growth (Figure 8), we can make some general conclusions about the overall successfulness of taxa used in the littoral community restoration. There appears to be two general categories, taxa that survived and/or expanded well (wild celery, water stargrass and bushy pondweed), and taxa that had lower survival and expansion rates (muskgrass, northern watermilfoil and flatstem pondweed).

The most successful taxa was water stargrass, with plants surviving in 100% of the transplanted sites in 2010 and 2011, and 59% of the sites in 2012 (Figure 5). Water stargrass had a mean homogenous area of 0.18 ± 0.04m^2 during the first year, and the largest homogenous area in 2011 of 3.12 ± 1.91m^2. Water stargrass also had one of the largest areas of presence with a mean area of 89.6 ± 24.8m^2 in 2011 and 38.7 ± 12.9m^2 in 2012 (Table 3).

Bushy pondweed also had good success, being found at 100% of the sites in 2010 and 92% of the sites in 2011, but only 33% of sites in 2012 (Figure 5). Mean homogenous area of bushy pondweed during the first year was 0.18 ± 0.03m^2, and 2.5 ± 0.9m^2 in 2011, but only 0.04 ± 0.02m^2 in 2012. Bushy pondweed had the largest area of presence in 2011 with 124.0 ± 61.2m^2, declining to 20.9 ± 13.0m^2 in 2012 (Table 3).

Wild celery also had good success being found at 100% of the sites in 2010, 92% of the sites in 2011, and 75% of sites in 2012 (Figure 5). Wild celery had a mean homogenous area of 0.4 ± 0.2m^2 in 2011 and increased to 1.3 ± 0.8m^2 in 2012. The mean area of presence of wild celery increased from 0.19 ± 0.15m^2 in 2010 to 1.4 ± 0.3m^2 in 2011 and to 3.5 ± 1.4m^2 in 2012 (Table 3). This shows that wild celery grew in more dense localized stands and didn't expand far beyond the original planted location until the third growing season (Table 3).

Northern watermilfoil had lower success with an initial survival of 92% in 2010, survival at 75% of sites in 2011, and survival at only 9% of sites in 2012 (Figure 5).

Northern milfoil had a very low homogenous area with $0.10 \pm 0.03m^2$ in all three years (Table 3). However, the area of presence for northern milfoil was considerably larger with $19.0 \pm 12.3m^2$ in 2011 but declined to $0.06 \pm 0.01m^2$ in 2012. In 2011, northern milfoil was found to have expanded reasonably well, however it didn't form a dense stand, rather it covered a broad area of individual stems. Northern watermilfoil was also highly affected by the early decrease in water clarity in 2012 with declines in both survival and area of presence.

Muskgrass was noted in 59% of the sites in 2010, 75% of the sites in 2011, and only 33% of sites in 2012 (Figure 5). Muskgrass had the smallest initial homogenous areas of only $0.04m^2$ in 2010. Expansion in the second and third growing seasons of muskgrass was poor, with a mean homogenous area of $0.26 \pm 0.15m^2$ in 2011 and $0.05 \pm 0.02m^2$ in 2012. It did however show a higher mean area of presence with $0.89 \pm 0.3m^2$ in 2011 and $8.12 \pm 5.33m^2$ in 2012 (Table 3). The growth characteristics of muskgrass resembled that of northern milfoil in that they generally occurred in small scattered patches.

Sediment Composition

Sediment organic matter ranged from 0.48% to 5.20% with a mean of 1.65% ± 0.25%. There was no significant difference in percent organic matter between open (1.62% ± 0.4%) and protected locations (1.68% ± 0.2%). Organic matter in the shallow locations (experiments one and two) was lower (1.62% ± 0.4%) than the deep transplant locations (experiments three and four) 1.92% ± 0.13%.

A linear mixed effect model showed no relationship of percent organic matter with plant homogenous area ($t = 0.874$, $n=12$), nor in organic matter versus area of presence ($t = -0.18$, $n=12$). Sediment bulk density was similar between shallow sampling locations ranging from 0.80g/ml to 1.97g/ml with a mean of 1.26g/ml ± 0.18 (Table 6). There was no significant relationship between open and protected locations with a protected location mean of 1.20g/ml ± 0.16 and open location mean of 1.31g/ml ± 0.18 (Table 6). The bulk density in deep transplant locations (0.72g/ml ± 0.04) was significantly lower than shallow sites. The linear mixed effect model showed no relationship between bulk density and plant coverage homogenous area ($t = -1.29$_$n=12$) or area of presence ($t = 0.47$_$n=12$).

The available sediment nitrogen showed relatively little variation between sampling locations with a mean of 0.037mg/g ± 0.0011(2se) (Table 6). This was likely due to several factors such as the consistent depth range from which the samples were taken. There was no significant difference between protected and open sites. Sediment nitrogen in the shallow locations was slightly lower compared to the deep transplant locations (experiment three and four) with deep locations having 0.005mg/g ± 0.001.

The linear mixed effect model showed the area of homogenous growth was significantly ($p< 0.05$) related to sediment nitrogen. Sites with high total nitrogen were the sites that failed to survive, and sites with a high area of homogenous growth had very low levels of total available nitrogen. These results likely reflect nitrogen uptake by plants rather than nitrogen being a deterrent of plant growth.

Water Clarity

There was an increase in springtime Secchi depth in 2010 and 2011 (Chapter II, Figure 7), after the removal of carp in 2009. However, the increased clarity did not persist for the duration of the summer. Summer Secchi depth dropped to less than 1.0m around mid-July in 2010, and early-August in 2011. Light PAR was measured at two sites per plot on 08 August 2011. There was no difference in light availability at sediment (0.5m) between open locations (1.8% ± 0.7) and protected (2.1% ± 0.6) locations.

Experiment 3: Transplanting depth, deeper plots planted in 2010

Plants failed to establish consistently at deeper depths. The plants all looked dead one month after planting. Evaluation of these plots in August 2011 found a few individual stems of flatstem pondweed at three of the four sites, and wild celery at one of the sites (Table 4), with no other transplanted species found. The 2012 assessments failed to find any of the transplanted taxa growing near any of the sites.

Experiment 4: Earlier planting of deeper plots, planted in 2011

The 2011 deeper transplants followed the trend of the 2010 deeper transplants in failing to thrive (Table 5). However the earlier date of deep transplanting in 2011 had some initial favorable results. Many of the flatstem pondweed and northern watermilfoil showed some initial growth and development of adventitious roots. Flatstem pondweed and wild celery both had a 67% survival rate, with at least one plant found in four of the six sites in August 2011. Although they were successful in surviving the summer, the average area of presence (0.01m^2 and 0.05m^2 respectively) was less than that which was planted in June (0.25m^2). This suggests while a few plants survived, most of them failed. After one month bushy pondweed was found growing in only one site, and while it had not expanded in coverage, the stems looked viable and healthy. None of the sites with muskgrass showed survival and plants became dark in color and brittle after the first month. During the spring assessment in 2012, wild celery was found to have a 33% survival rate, with a single rosette growing at two locations. However summer assessments failed to locate any transplanted species.

The date at which the 5% surface light threshold stayed below the 1.4 m depth (average depth of the deep transplants) can indicate the length of time light was available to the transplanted propagules. In 2010, the 5% threshold was met through the end of July; in 2011 the 5% threshold was crossed in early August. However in 2012, the 5% threshold was crossed in early June (Chapter II, Table 5).

Discussion:

To increase the effectiveness of transplanting submersed aquatic macrophytes several factors such as water clarity, life history traits, perennial structure size, and planting depth should be considered. Taxa with large perennial structures had survival rates greater than 80%, whereas annuals selected for transplanting had 50% or less survival. Taxa that use fragmentation strategies for reproduction showed the highest expansion. Light availability was the largest factor in survival of taxa, where transplants in shallow locations had higher survival than those panted in deeper locations.

Overwintering survival was a better predictor of establishment than survival during the initial growing season. Transplanting submersed aquatic macrophytes was shown to be effective in shallow lakes of northern climates after common carp populations have been reduced.

Water stargrass and wild celery, both with large perennial structures, exhibited the highest survival in this case study. Their survival rate was similar between experiment one and two. This is consistent with Madsen (1991) in that taxa with large vegetative structures have larger carbohydrate reserves, allowing for faster initial growth phase and were generally sufficient to overwinter. He also noted that perennial submersed macrophytes often produce twice as many vegetative propagules as seeds and vegetative propagules are generally better at overwintering than seeds. Measuring success by taking into consideration both survival rate and expansion rate, transplanted taxa generally fell into two categories those with fairly high success and those with relatively low success (Figure 8). Taxa with annual life history traits such as muskgrass, flatstem pondweed and northern watermilfoil had poor survival (Figure 8). The primary means of overwintering by annual plants is through the production of seeds. While seed producers were noted as having lower survival rates, seed production can be a better means of broadcasting propagules greater distances (Madsen 1991). This explains why bushy pondweed, an annual with mixed survival, can be considered to relatively successful; where it did survive it generally grew and expanded well.

Life history traits were not a predictor of expansion like they were for establishment. Factors for predicting expansion were not clear. Taxa using fragmentation strategies, such as water stargrass and bushy pondweed, had the highest expansion (Figure 8). Water stargrass had good survival and most sites had high expansion. The use of stolons and runners by water stargrass likely lead to increased success, as this was also noted in Smart et al., (1996). Wild celery had very good overwinter survival each year but little expansion. This result was unexpected as wild celery was noted to have expanded rapidly in Smart et al., (1998). None of the transplanted taxa developed large dense homogenous stands. When the transplants expanded beyond their original planted areas they generally grew in scattered low density clusters.

Light availability was the largest factor in survival of all taxa. Transplants in shallow locations with higher available light resulted in high survival rates. All deeper sites failed to survive past one year with the exception of two individual wild celery and two flatstem pondweed plants (taxa with the largest perennial structures). The subsequent failure of the deeper sites (mean depth 1.4m) was due to poor water clarity and low light availability during the mid-summer. When the plants were transplanted in deeper locations (1.4m) on 22 July 2010 the Secchi disk depth was only 1.0m and it dropped to 0.5m within three weeks. Even though the plants were mature, approximately 0.75m in height, they were unable to effectively adapt to the lower light levels. Light compensation point helps explain reduced survival in lower clarity. Madsen (1991) found an average of 7% surface light was needed across species, however light compensation point varies between species. Wild celery plants receiving 5% surface light, for more than 95 days, were able to produce tubers (Kimber 1995), however, at 2% light plants failed. Sand-Jensen and Madsen (1991) found plants formed new tissue and elongated slightly because of photosynthesis and reallocation, but this was balanced by weight losses in other tissues (presumably reproductive structures). Root development also decreases with decreasing light availability (Sand-Jensen and Madsen 1991). Survival of all taxa from both experiments decreased during 2012. Unseasonably high air temperatures in early spring of 2012 caused a rapid increase in water temperatures and subsequent algae blooms. This resulted in decreased water clarity and Secchi depths remained less than 0.5m from early June 2012 throughout the summer. Light availability, being one of the most important environmental variables for submersed macrophytes (Scheffer 1998) likely explains the decrease in both survival and expansion of the transplants during the 2012 summer.

Floating leaf lily stands do not provide adequate protection to substantially affect the outcomes of transplanting. Protection behind lily beds may not have been significantly different because of the small size of the study lake. During the first season of experiment one, the caged transplants showed slightly better growth, but little difference in survival compared to open transplants. However, there was no difference in growth or survival between caged and open transplants the next spring, indicating that overwintering survival played a larger role in establishment than cage protection. The caged plots did not appear to have the same increase in growth as seen by other studies. Lauridsen et al., (2003) found that protective barriers were most effective in the spring and immediately after transplanting and less effective after midsummer. Protective barriers may not have been as beneficial on Lake Susan because the carp abundance was greatly reduced, therefore were few benthivorous fish to uproot plants. The density of other macro-herbivores such as waterfowl or muskrats was not quantified, though personal observation indicated very low abundance of each.

Monitoring transplant success should be done over multiple years, not just within the initial growing season. All the transplants generally showed good survival during the initial growing season. Overwintering survival however, was more varied and appeared to be a better indicator of establishment. Plants that overwintered generally established and expanded in the subsequent years. Understanding long term survival and expansion of selected taxa can increase efficiency of future littoral restorations. The rate of planting should be adjusted for establishment and expansion. Taxa with large perennial structures may be more flexible at greater depths. Annuals planted in shallow locations may have inconsistent survival, but higher expansion.

Further research is needed to provide greater understanding on the factors that better predict expansion and colonization of the littoral zone. More focus on timing interactions may be beneficial for increasing the successfulness of transplanting in deeper areas. The earlier date of deep transplanting in 2011 had some initial favorable results. Many of the flatstem pondweed and northern watermilfoil showed growth and development of adventitious roots. By August 2011, water clarity was poor and the majority of plants failed to survive. Earlier planting may allow more time to overcome transplant shock and develop overwintering storage in more favorable light conditions. Size of plant or perennial structure can influence plant health and ability to withstand transplant shock (Zimmerman et al. 1995), however the size of transplants were not measured.

In conclusion, this case study adds to the body of literature supporting transplanting submersed aquatic macrophytes as a means of restoring the littoral community (Dick et al. 2004, Smiley and Dibble 2006, and Storch et al. 1986), and shows these methods are successful in northern climates. Before an aquatic macrophyte restoration is attempted the ecological stressors that prohibit the natural recruitment of macrophyte, the population of carp in this case, should first be addressed (Smart et al. 2005). The maximum depth of macrophyte transplants generally should not be greater than the mid-summer Secchi depth, as the lower light available will likely result in reduced survival. Broader scale success of littoral community restoration is dependent on sustained improvements in water clarity.

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Tables Chapter III

Table 1. Experiment one comparison of survival and mean plant height of transplanted taxa at caged vs. open locations at the end of the initial season, September 2009. Mean height and depth calculated with only successful sites.

		Survival	Mean Depth (cm)	Mean Height (cm)
Muskgrass	caged	63%	40	22.4
	open	50%	43.5	22.6
N. watermilfoil	caged	100%	41	64.0
	open	50%	42.5	34.5
Bushy pondweed	caged	100%	42	63.4
	open	100%	42.5	61.9
Wild celery	caged	88%	42.5	32.1
	open	75%	43	14.4
Water stargrass	caged	100%	40	53.5
	open	88%	43.5	46.9

Table 2. Experiment one summary of survival, height, and area of presence ± one standard error, of transplanted taxa at the end of the third growing season, August 2011. Mean height and area were calculated with only successful sites.

	Survival	Mean height (cm)	Mean Area of presence (m^2) all sites
Muskgrass	56%	32.3 ± 4.7	0.28 ± 0.25
N. watermilfoil	6%	0.0 ± 0.0	0.0 ± 0.0
Bushy pondweed	6%	50.0 ± 0.1	8.29 ± 11.86
Wild celery	94%	57.3 ± 7.2	16.0 ± 18.2
Water stargrass	81%	57.5 ± 5.8	35.8 ± 19.8

Table 3. Experiment two mean homogenous area (m^2) and mean area of presence (m^2) transplants. Means calculated for only successful sites. Error values indicate one standard error.

Mean homogenous area (m^2)

	2010	2011	2012
Muskgrass	0.04 ± .02	0.26 ± 0.15	0.05 ± 0.02
N. watermilfoil	0.10 ± 0.03	0.01 ± 0.0	0.06 ± 0.01
Bushy pondweed	0.18 ± 0.03	2.05 ± 0.98	0.04 ± 0.02
Wild celery	0.19 ± 0.15	0.36 ± 0.15	1.31 ± 0.84
Water stargrass	0.18 ± 0.04	3.12 ± 1.91	0.04 ± 0.02

Mean area of presence (m^2)

	2010	2011	2012
Muskgrass	0.04 ± .02	0.89 ± 0.26	8.12 ± 5.33
N. watermilfoil	0.10 ± 0.03	18.97 ± 12.28	0.06 ± 0.01
Bushy pondweed	0.18 ± 0.03	124.05 ± 61.2	20.95 ± 13.01
Wild celery	0.19 ± 0.15	1.43 ± 0.34	3.52 ± 1.42
Water stargrass	0.18 ± 0.04	89.58 ± 24.81	38.72 ± 12.99

Table 4. Summary of 2010 survival, height, and growth of transplanted taxa in experiment three. Mean height calculated from only successful sites, mean area of presence calculated with all sites, successful and failed together.

	Survival	Height (cm)	Area of Presence (m^2)
Muskgrass	0%	0.0	0.00
N. watermilfoil	0%	0.0	0.00
Wild celery	25%	40	0.01
Bushy pondweed	0%	0.0	0.00
Flatstem pondweed	75%	50	0.03

Table 5. Summary of survival 2011, height, and growth of transplanted taxa in experiment four. Mean height calculated from only successful sites, mean area of presence calculated with all sites, successful and failed together.

	Survival	Height (cm)	Area of Presence (m^2)
Muskgrass	0%	0.0	0.000
N. watermilfoil	0%	0.0	0.000
Wild celery	67%	43.5	0.049
Bushy pondweed	17%	8.3	0.003
Flatstem pondweed	67%	53.3	0.006

Table 6. Summary of mean sediment bulk density, mean percent organic matter and mean total nitrogen in soil samples taken July 2011. Range given in 2 standard errors.

	Open (Shallow)	Protected (Shallow)	Combined (Shallow)	Deep
Density (g/ml)	1.31±0.18	1.21±0.17	1.26±0.18	0.72±0.04
Organic (%)	1.62±0.80	1.68±0.40	1.66±0.49	1.92±0.25
Total N mg/g	0.0036±0.0009	0.0039±0.0020	0.0037±0.0011	0.0053±0.0008

Figures Chapter III

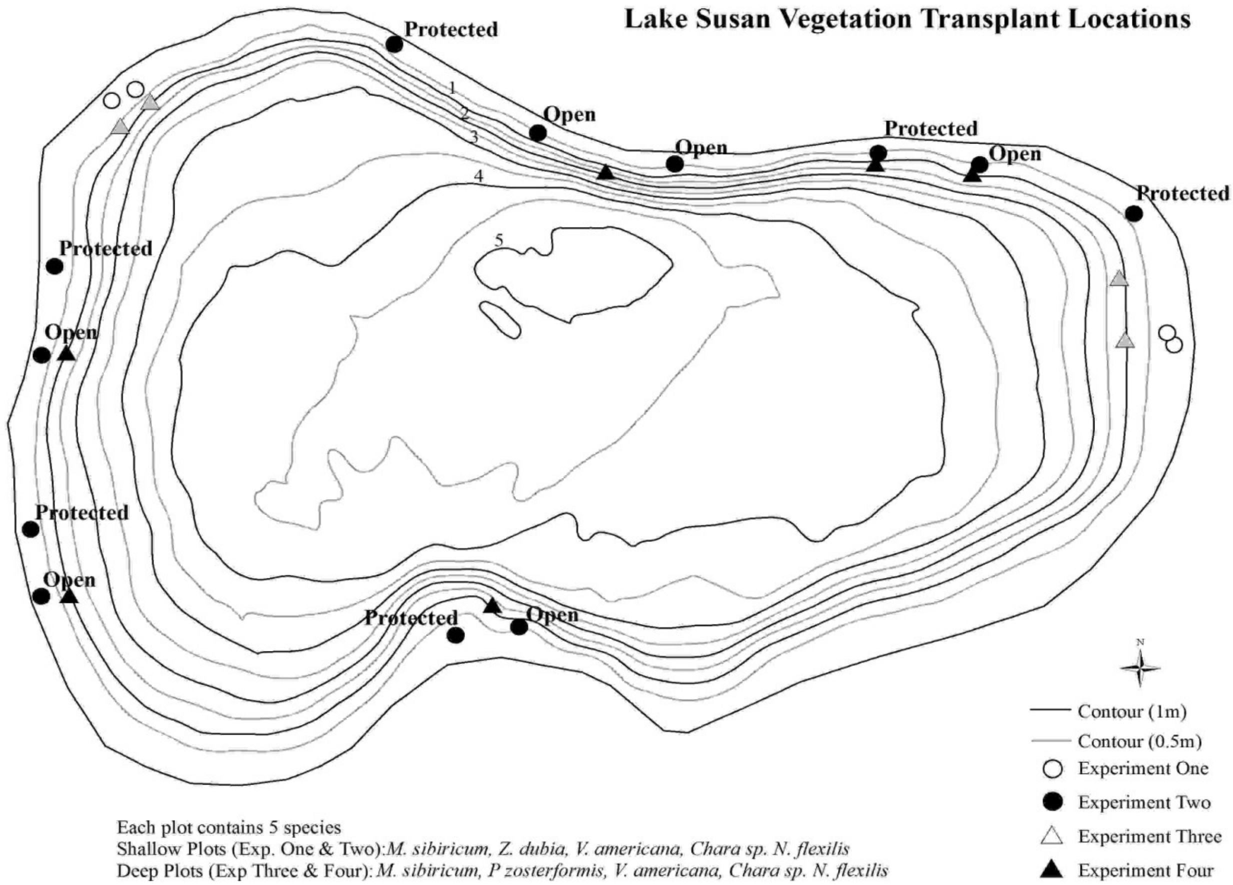

Figure 1. Locations of transplant plots in Lake Susan. Each plot contained five sites with one species planted at each site.

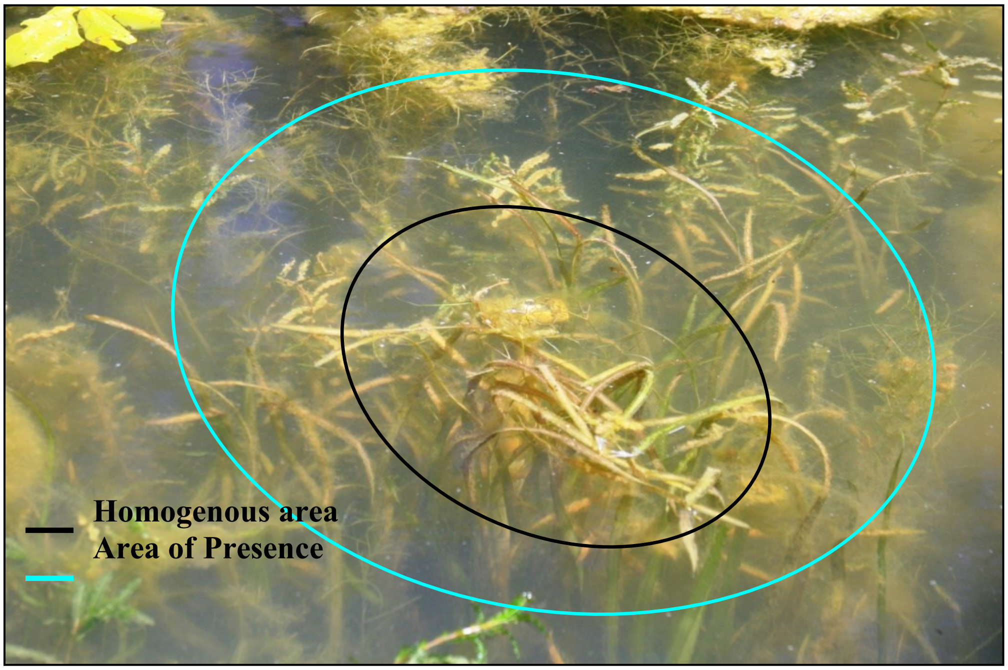

Figure 2. Example of plant growth assessment with area of presence and homogenous area of growth indicated in experiment two. This site was planted with wild celery (*Vallisneria americana*).

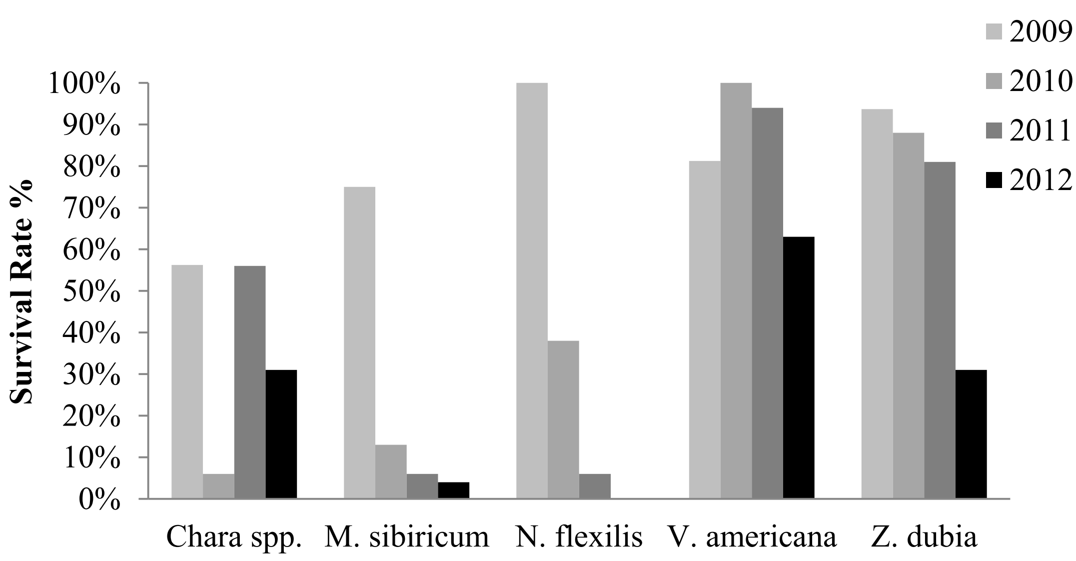

Figure 3. Experiment one survival (percent of sites with taxa present) of transplanted taxa by year.

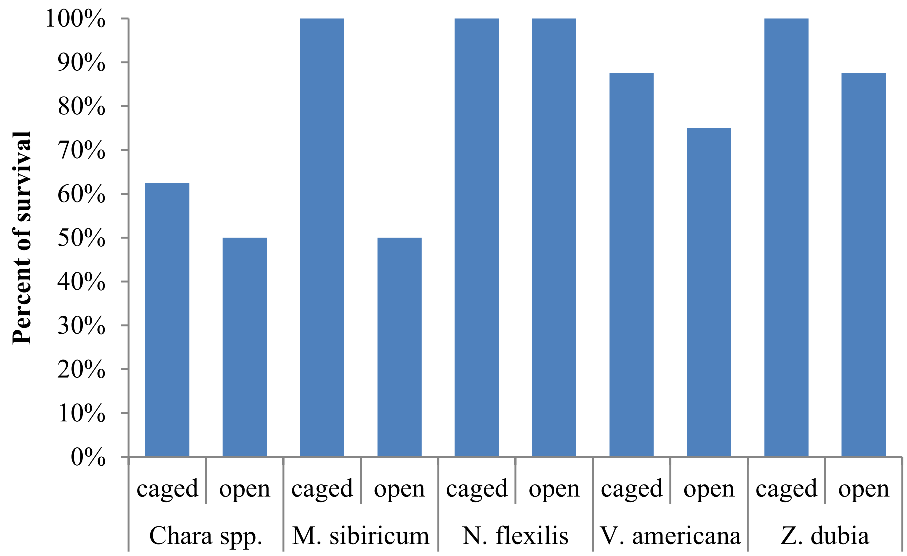

Figure 4. Experiment one survival rate comparison of transplants in caged vs. open location during the initial growing season (2009).

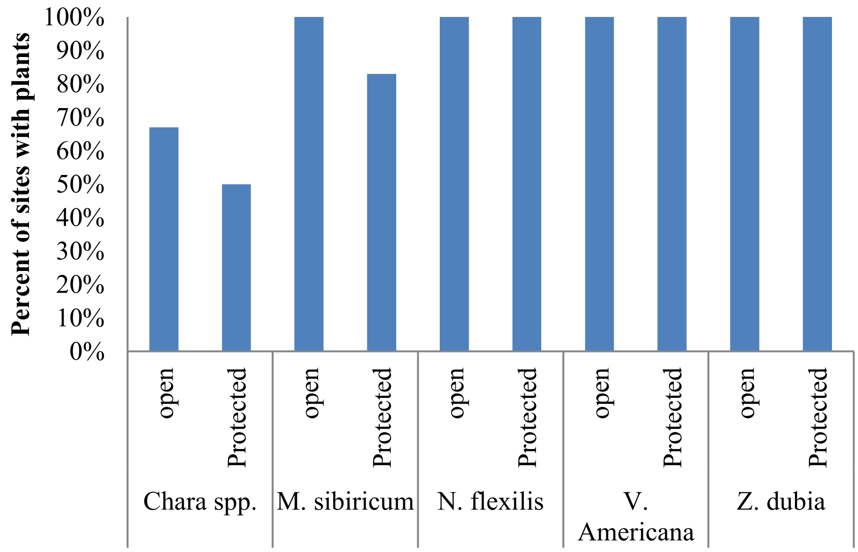

Figure 5. Experiment two survival of transplanted taxa in open versus protected sites (behind lilies) in 2010.

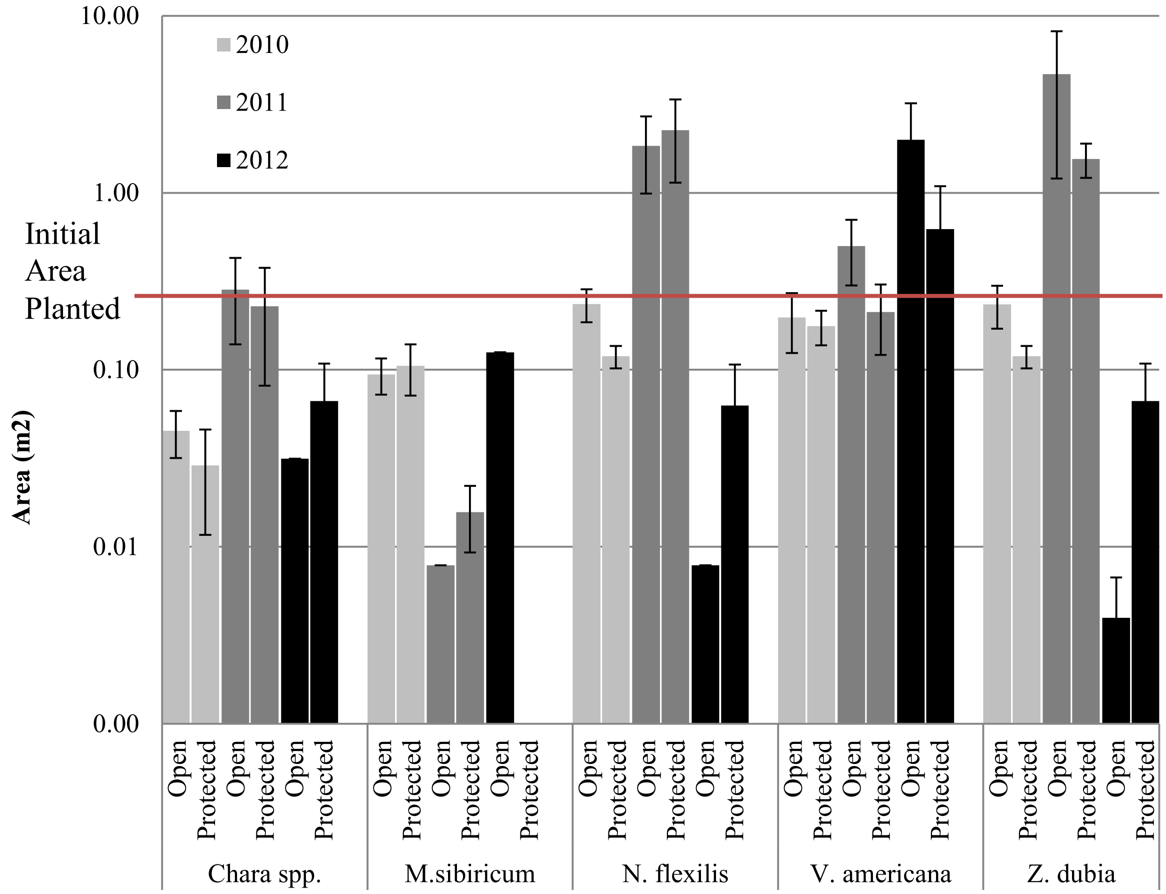

Figure 6. Homogenous area (m^2) of successful sites by year of experiment two, plants transplanted August 2010. Means calculated from successful sites only. Area axis is in logarithmic scale. Error bars represent one standard error.

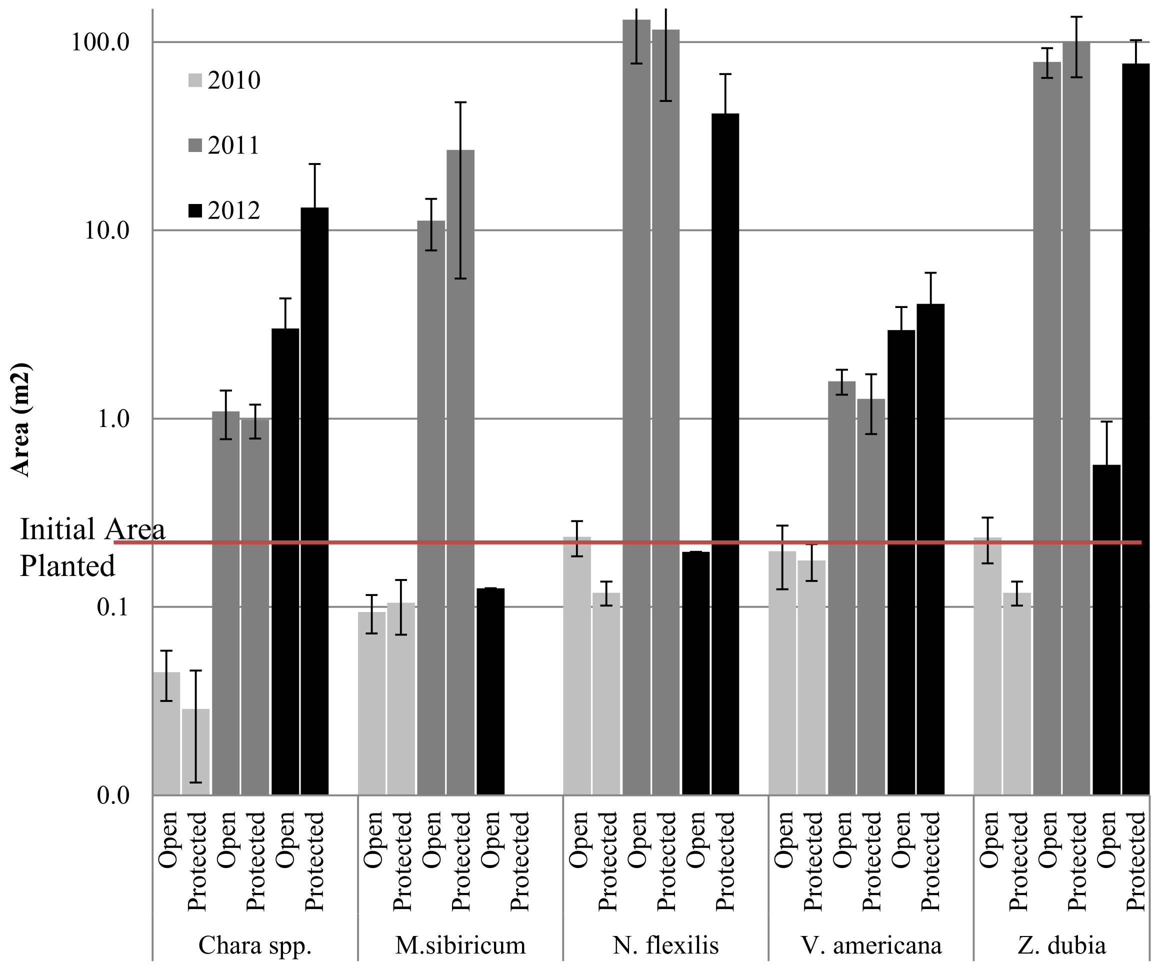

Figure 7. Comparison of mean area of presence (m^2) of transplants in open versus protected sites in experiment two, plants transplanted August 2010. Means calculated from successful sites only. Area axis is in logarithmic scale. Error bars represent one standard error.

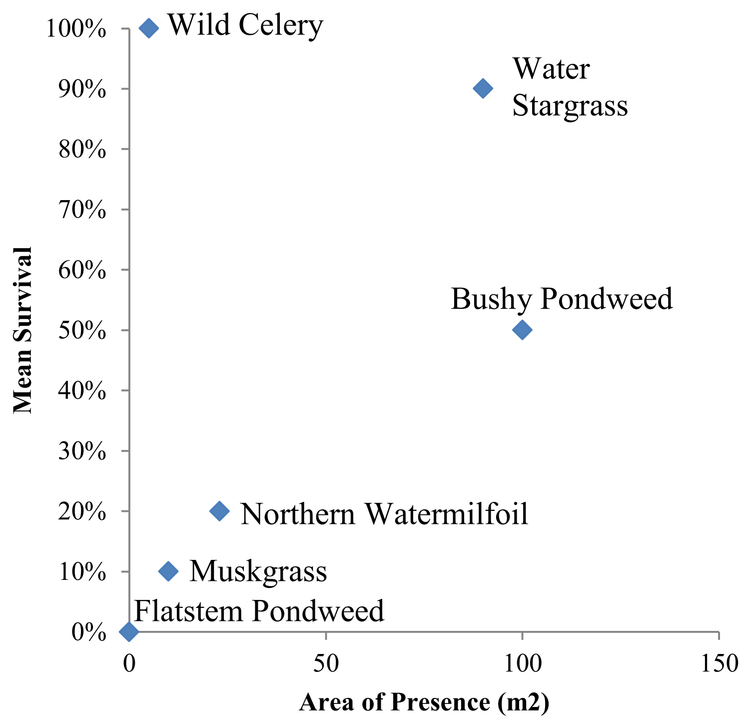

Figure 8. Comparison of mean survival and area of presence of taxa in experiment 2 during the 2011 growing season.

Chapter IV

Concluding Remarks:

Aquatic macrophytes play a significant role in maintaining water quality and ecosystem health. Aquatic plants reduce wave energy and rooted plants directly reduced re-suspension of sediment. Aquatic plants compete with algae for the uptake of suspended nutrients, as well as provide structure for epiphytes and zooplankton. The reduction of available nutrients in the water column creates a positive feedback loop to help maintain water clarity, and further enhancing the aquatic plant community (Carpenter and Lodge 1986, Scheffer et al. 2001). High densities of common carp uproot aquatic macrophytes and stir up sediments, encouraging a turbid water state (Crivelli 1983, Hanson and Butler 1994, Scheffer 1998). Lakes not impacted by high carp populations often have higher plant diversity. Lake Ann for example, just 3km upstream from Lake Susan, had low abundance of carp and 22 species of aquatic plants. Reducing the common carp population in turbid lakes that have a very high abundance of carp, can restore the aquatic macrophytes and improve water quality (Bajer et al. 2009).

I assessed the response of the aquatic macrophyte community for four years after the large scale removal of common carp from Lake Susan, Carver County, MN. I found a positive response in the abundance and distribution of both native and non-native submersed aquatic macrophytes in Lake Susan after the reduction of the common carp population. These results support those found by Crivelli (1983), Hanson and Butler (1994), Scheffer (1998), and Bajer et al., (2009). However, there was only a slight increase in species richness as only two new taxa recruited naturally. The reductions in carp abundance lead to an increase in the distribution and density of aquatic plants to the point where light availability became the limiting factor. The non-native species curlyleaf pondweed increased significantly in frequency and density in just three years after carp removal. This increase of a non-native was similar to that noted by Lauridsen et al., (1994) after a large scale fish bio-manipulation. The other non-native taxa, Eurasian watermilfoil, showed an annual decrease in frequency of occurrence and biomass between 2009 and 2012. This, combined with the persistent high density of milfoil weevils, suggests the weevils were controlling the Eurasian watermilfoil.

There are two likely mechanisms behind the increase in frequency and density of aquatic macrophytes. The biggest factor was likely the reduction in direct uprooting of plants by carp. The other factor in increasing the plant community was the increase in spring time and early summer clarity. However, few new taxa recruited in the lake, at least beyond the initial year after removal. Manipulation of the littoral community, such as transplanting whole plants, may be necessary to promote the establishment and distribution of native vegetation.

I assessed and compared the survival and expansion of six transplanted submersed aquatic macrophyte taxa around the littoral zone. Over the course of the three year study period there was favorable survival and expansion of many of the transplants There was general agreement between our case study and the findings of Smart et al., (1996). There was high initial survival of most transplanted taxa in shallow sites (< 1m). For many taxa, overwintering success was a better predictor of long-term survival, than was initial survival. Over three years, wild celery showed high survival but rather limited expansion, whereas water stargrass showed good survival and higher expansion. Bushy pondweed and muskgrass has lower survival and variable growth rates, and northern watermilfoil survival decreased each year. Transplants were not successful at greater depths. Most plants transplanted in deeper sites (>1.4m) showed poor initial survival and only a very few plants successfully overwintered. By the end of the second growing season all the plants had failed. This is likely related to the low summer light availability at depths >1m, when summer Secchi depths drop below 1m. While earlier planting resulted in better growth during the initial season, the transplanted taxa largely failed to survive overwinter.

To succeed over winter the aquatic plants need to have enough stored energy, either by carbohydrate storage in the roots or by seed. Success and expansion over the following growing season is likely related to the plants photosynthesis ability, thus clarity related. The success of water stargrass and wild celery is likely aided by their ability to store carbohydrates in tubers, whereas musk grass relies on the production and success of spores.

Improvements in the water clarity during the spring did not persist through the summer in Lake Susan after the reduction of carp abundance. This was likely caused by internal loading of phosphorous. As surface water temperatures increased so too did thermal and oxygen stratification. The hypolimnion became anoxic during the summer and likely released sediment bound phosphorous, driving algae blooms and decreasing water clarity.

I found little difference in success at locations that were protected from herbivore access, and locations open to herbivore access. This does not directly contradict Lauridsen et al., (2003) who found protection from herbivory important for successful transplants, because of the very low levels of benthivorous fish (carp were removed), and relatively few waterfowl in Lake Susan. There was also no difference in survival or growth between transplants located shoreward of floating leaf lilies and open to full wave action.

This thesis has shown that both native and exotic submersed aquatic plants increase in frequency and density in response to the removal of high density of common carp. I have also shown that transplanting whole plants from a neighboring lake can be an effective means to increase species composition and richness of submersed aquatic plants in northern latitude lakes. Light availability was shown to be an important factor in survival and expansion of the transplants. Thus minimum summertime water clarity must be considered in determining planting depth of transplants and more broadly to reestablish native submersed aquatic macrophytes.

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